Reptile Conservation

Global evidence for the effects of interventions for reptiles

Katherine A. Sainsbury, William H. Morgan, Maggie Watson, Guy Rotem, Amos Bouskila, Rebecca K. Smith & William J. Sutherland

CONSERVATION EVIDENCE SERIES SYNOPSES

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Conservation Evidence Series Synopses

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Cover image: Borneo forest dragon (*Gonocephalus borneensis*) in Kubah National Park, Malaysia. Photograph by John White.

Digital material and resources associated with this synopsis are available at
https://www.conservationevidence.com/

Contents

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1. About this book

The Conservation Evidence project

The Conservation Evidence project has four main parts:

1. The **synopses** of the evidence captured for the conservation of particular species groups or habitats, such as this synopsis. Synopses bring together the evidence for each possible intervention. They are freely available online and, in some cases, available to purchase in printed book form.

2. An ever-expanding **database of summaries** of previously published scientific papers, reports, reviews or systematic reviews that document the effects of interventions. This resource comprises over 7,800 pieces of evidence, all available in a searchable database on the website www.conservationevidence.com.

3. ***What Works in Conservation,*** which is an assessment of the effectiveness of interventions by expert panels, based on the collated evidence for each intervention for each species group or habitat covered by our synopses. This is available as part of the searchable database and is published as an updated book edition each year (www.conservationevidence.com/content/page/79).

4. An online, **open access journal** *Conservation Evidence* publishes new pieces of research on the effects of conservation management interventions. All our papers are written by, or in conjunction with, those who carried out the conservation work and include some monitoring of its effects (www.conservationevidence.com/collection/view).

The purpose of Conservation Evidence synopses

Conservation Evidence synopses **do**	Conservation Evidence synopses **do not**
• Bring together scientific evidence captured by the Conservation Evidence project (over 7,800 studies so far) on the effects of interventions to conserve biodiversity	• Include evidence on the basic ecology of species or habitats, or threats to them
• List all realistic interventions for the species group or habitat in question, regardless of how much evidence for their effects is available	• Make any attempt to weight or prioritize interventions according to their importance or the size of their effects

- Describe each piece of evidence, including methods, as clearly as possible, allowing readers to assess the quality of evidence
- Work in partnership with conservation practitioners, policymakers and scientists to develop the list of interventions and ensure we have covered the most important literature
- Weight or numerically evaluate the evidence according to its quality
- Provide recommendations for conservation problems, but instead provide scientific information to help with decision-making

Who this synopsis is for

If you are reading this, we hope you are someone who has to make decisions about how best to support or conserve biodiversity. You might be a land manager, a conservationist in the public or private sector, a farmer, a campaigner, an advisor or consultant, a policymaker, a researcher or someone taking action to protect your own local wildlife. Our synopses summarize scientific evidence relevant to your conservation objectives and the actions you could take to achieve them.

We do not aim to make your decisions for you, but to support your decision-making by telling you what evidence there is (or isn't) about the effects that your planned actions could have.

When decisions have to be made with particularly important consequences, we recommend carrying out a systematic review, as the latter is likely to be more comprehensive than the summary of evidence presented here. Guidance on how to carry out systematic reviews can be found from the Centre for Evidence-Based Conservation at the University of Bangor (www.cebc.bangor.ac.uk).

Background

At present, some 11,440 extant reptile species have been described on Earth and several hundred new species have been described each year since 2008 (Uetz & Hosek 2018). As grazers, seed dispersers, predators, prey and commensal species, reptiles perform crucial functions in ecosystems (Böhm *et al*. 2013).

Reptiles are a hugely diverse group of animals (Pincheira-Donoso *et al*. 2013) and are adapted to live in a wide range of tropical, temperate and desert terrestrial habitats, as well as freshwater and marine environments (Böhm *et al*. 2013). That said, reptile species usually have narrower geographic distributions than other vertebrate taxonomic groups (e.g. birds or mammals), and this coupled with particular life history traits makes some reptile species particularly vulnerable to anthropogenic threats (Böhm *et al*. 2013, Fitzgerald *et al*. 2018). For example, some turtle species are typically very long lived, take years to reach full maturity, produce small clutches and have variable reproductive success, which means that they are vulnerable to loss of adults and take many years to recover from declines (Congdon *et al*. 1994).

Multiple threats to reptile populations have been identified and are implicated in species declines (Gibbons *et al*. 2000, Todd *et al*. 2010). These threats include habitat modification, loss and fragmentation (Neilly *et al.* 2018, Todd *et al.* 2017), environmental contamination (Sparling *et al. 2010*), potentially unsustainable harvesting and/or collection (van Cao *et al.* 2014), invasive species (Fordham *et al.* 2006), climate change (Bickford *et al.* 2010, Sinervo *et al.* 2010) and disease and parasitism (Seigel *et al.* 2003). Also, due to their physical characteristics, reputation (warranted or otherwise) and in some cases venomous bites, some reptile species are viewed with distaste, which leads to apathy around their conservation (Gibbons *et al*. 1988). According to the IUCN Red List, of 10,148 reptile species that have been assessed, some 21% are considered to be threatened (IUCN 2021). Extinction risks are particularly high in tropical regions, on oceanic islands and in freshwater environments (Böhm *et al*. 2013), with some 59% of turtle species assessed at risk of extinction (van Dijk *et al*. 2014). Reptiles with specialist habitat requirements and limited ranges that are in areas accessible to humans are likely to face greater extinction risks (Böhm *et al*. 2016). Many island reptile species are endemic and are therefore even more vulnerable to extinction as a result of human disturbance (Fitzgerald *et al*. 2018). For a comprehensive summary of threats to different families of reptiles see Fitzgerald *et al*. (2018).

Evidence-based knowledge is key for planning successful conservation strategies and for the cost-effective allocation of scarce conservation resources. To date, reptile conservation efforts have involved a broad range of actions, including protection of eggs, nests and nesting sites; protection from predation; translocations; captive breeding, rearing and releasing; habitat protection, restoration and management; and addressing the threats of accidental and intentional harvesting. However, most of the evidence for the effectiveness of these interventions has not yet been synthesised within a formal review and those that have could benefit from periodic updates in light of new research.

Targeted reviews are labour-intensive and expensive. Furthermore, they are ill-suited for subject areas where the data are scarce and patchy. Here, we use a subject-wide evidence synthesis approach (Sutherland *et al*. 2019) to simultaneously summarize the evidence for the wide range of interventions dedicated to the conservation of all reptiles. By simultaneously targeting all interventions, we are able to review the evidence for each intervention cost-effectively, and the resulting synopsis can be updated periodically and efficiently. The synopsis is freely available at www.conservationevidence.com and, alongside the Conservation Evidence online database, is a valuable asset to the toolkit of practitioners and policy makers seeking sound information to support reptile conservation. We aim to periodically update the synopsis to incorporate new research. The methods used to produce the Reptile Conservation Synopsis are outlined below.

Scope of the Reptile Conservation synopsis

Review subject

This synthesis focuses on global evidence for the effectiveness of interventions for the conservation of reptiles. This subject has not yet been covered using subject-wide evidence synthesis. This is defined as a systematic method of reviewing and synthesising evidence that covers broad subjects (in this case conservation of multiple taxa) at once, including all closed review topics within that subject at a fine scale, and analysing results through study summary and expert assessment, or through meta-analysis. The term can also refer to any product arising from this process (Sutherland *et al*. 2019). This global synthesis collates evidence for the effects of conservation interventions on terrestrial, aquatic and semi-aquatic reptiles, including all reptile orders, i.e. Crocodilia (alligators, crocodiles and gharials), Testudines (turtles and tortoises), Squamata (snakes, lizards and amphisbaenians) and Rhynchocephalia (tuatara). This synthesis covers evidence for the effects of conservation interventions for wild reptiles (i.e. not in captivity). We have not included evidence from the substantial literature on husbandry of marine and freshwater reptiles kept in zoos or aquariums. However, where these interventions are relevant to the conservation of wild declining or threatened species, they have been included, e.g. captive breeding for the purpose of increasing population sizes (potentially for reintroductions) or gene banking (for future release).

For this synthesis, conservation interventions include management measures or interventions that aim to conserve wild reptile populations and reduce or remove the negative effects of threats. The output of the project is an authoritative, transparent, freely accessible evidence-base of summarized studies and expert assessment scores that will support reptile management decisions and help to achieve conservation outcomes.

Advisory board

An advisory board made up of international conservationists and academics with expertise in terrestrial and aquatic reptile conservation was formed. These experts inputted into the evidence synthesis at three key stages: a) identifying key sources of evidence, b) developing a comprehensive list of conservation interventions for review and c) reviewing the draft evidence synthesis. The advisory board is listed above.

Creating the list of interventions

At the start of the project, a comprehensive list of interventions was developed by searching the literature and in partnership with the advisory board. The list was also checked by Conservation Evidence to ensure that it followed the standard structure. The aim was to include all interventions that have been carried out or advised to support populations or communities of wild reptiles, whether evidence for the effectiveness of an intervention is available or not. During the synthesis process further interventions were discovered and integrated into the synopsis structure. The list of interventions is organized into categories based on the IUCN classifications of direct threats (www.iucnredlist.org/resources/threat-classification-scheme) and conservation actions (www.iucnredlist.org/resources/conservation-actions-classification-scheme). For interventions with a large body of literature, the intervention may be split into different methods of implementation (e.g. different designs, implementation in different seasons, different methods for acclimatisation before release etc.), different species/functional groups, or broad habitats, if relevant to do so.

In total, we found 242 conservation and/or management interventions that could be carried out to conserve marine and freshwater reptile populations. We found evidence for the effects on terrestrial and aquatic reptile populations for 189 of these interventions. The evidence was reported as 959 summaries from 676 relevant publications found during our searches (see Methods below).

Methods

Literature searches

Literature was obtained from the Conservation Evidence discipline-wide literature database, and from searches of additional subject specific literature sources (see Appendices 1–2). The Conservation Evidence discipline-wide literature database is compiled using systematic searches of journals (all titles and abstracts) and report series ('grey literature'); relevant publications describing studies of conservation interventions for all species groups and habitats were saved from each and were added to the database. Final lists of evidence sources searched for this synopsis are published in this synopsis document (see Appendices 1–2), and the full list of journals and report series is published online (www.conservationevidence.com/journalsearcher/synopsis).

a) Global evidence

Evidence from all around the world was included.

b) Languages included

The following journals that included articles in German and Spanish were searched and relevant papers extracted:

- Herpetozoa (1988–2018)
- Revista de Biologia Tropical (1976–2018)

All other journals searched are published in English or at least carry English summaries. All relevant papers were added to the Conservation Evidence discipline-wide literature database (see below).

c) Journals searched

i) From the Conservation Evidence discipline-wide literature database

All journals (and years) listed in Appendix 1b were searched, and relevant papers added to the Conservation Evidence discipline-wide literature database. An asterisk indicates the journals most relevant to this synopsis. Others are less likely to have included papers relevant to this synopsis, but if they did, they were summarized.

ii) Update searches

Additional searches up to the end of 2018 were undertaken by the synopsis authors for journals likely to yield studies for reptiles (see Appendix 1a, journals marked with asterisks).

iii) New searches

In addition to those above, new focused searches of journals relevant to the conservation of reptile populations were undertaken by the synopsis authors (indicated in bold Appendix 1a). These journals were identified through expert judgement by the project researchers and the advisory board and ranked in order of relevance, to prioritise searches that were considered likely to yield higher numbers of relevant studies.

- Asian Herpetological Research (2010–2018)
- Asiatic Herpetological Research (1993–2008)
- Basic and Applied Herpetology (2011–2018)
- Bibliotheca Herpetologica (1999–2017)
- Bulletin of the Maryland Herpetological Society (1980–2015)
- Caribbean Herpetology (2010–2018)
- Chelonian Conservation and Biology (1993–1996 & 2005–2018)
- Chelonian Research Monographs (1996–2017)
- Collinsorum (formerly Journal of Kansas Herpetology) (2002–2018)

- Herpetological Review (1967–2018)
- Herpetology Notes (2008–2018)
- Herpetozoa (1988–2018)
- Journal of North American Herpetology (2014–2017)
- Kansas Herpetological Society Newsletter (1977, 1983, 1998 & 2001)
- Mesoamerican Herpetology (2014–2017)
- Phyllomedusa (2002–2018)
- Testudo (1978–2017)

A number of journals were searched, but relevant studies not included in the synopsis due to time constraints or access restrictions. These journals are:

- Biawak (2007-2017)
- Bulletin of the Chicago Herpetological Society (1990-2018)
- Journal of Herpetological Medicine and Surgery (2000-2018)
- Russian Journal of Herpetology (1996-2018)
- Salamandra (1980-2018)

d) Reports from specialist websites searched

i) From the Conservation Evidence discipline-wide literature database

All report series (and years) in Appendix 2b were searched for the Conservation Evidence project. An asterisk indicates the report series most relevant to this synopsis. Others are less likely to have included reports relevant to this synopsis, but if they did, they were summarized.

ii) Update searches

Updated searches of report series already searched as part of the wider Conservation Evidence project were not undertaken for this synopsis.

iii) New searches

New searches targeted specialist reports relevant to reptile conservation as listed in Appendix 2a. These searches reviewed every report title and abstract or summary within each report series (published before the end of 2018) and added any relevant report to the project database.

A number of reptile report series were searched but the findings were not summarized due to time constraints:

- African Sea Turtle Newsletter (2014-2018)
- Marine Turtle Newsletter (1976-2018)

- Reptile Rap (1999–2016)

e) Other literature searches

The online database www.conservationevidence.com was searched for relevant publications that have already been summarized.

Where a systematic review was found for an intervention, then only the systematic review was summarized. Non-systematic reviews (or editorial, synthesis, preface, introduction etc.) that provided new/collective data were included/summarized (but the relevant publications referenced within it were not summarized individually). Relevant publications cited in other publications summarized for the synopsis, were not included/summarized (due to time constraints).

f) Supplementary literature identified by advisory board or relevant stakeholders

Additional journal or specialist website searches, and relevant papers or reports suggested by the advisory board or relevant stakeholders were also included, if relevant.

g) Search record database

A database was created of all relevant publications found during searches. Reasons for exclusion were recorded for all those included during screening that were not summarized for the synopsis.

Publication screening and inclusion criteria

A summary of the total number of evidence sources and papers/reports screened is presented in the diagram in Appendix 3.

a) Screening

To ensure consistency/accuracy when screening publications for inclusion in the literature database, an initial test using the Conservation Evidence inclusion criteria (provided below) and a consistent set of references was carried out by authors, compared with the decisions of the experienced core Conservation Evidence team. Results were analysed using Cohen's Kappa test (Cohen 1960). Where initial results did not show 'substantial' (K = 0.61–0.8) or 'almost perfect' agreement (K = 0.81–1.0), authors were given further training. A second Kappa test was used to assess the consistency/accuracy of article screening for the first two years of the first journal searched by each author. Again, where results did not show 'substantial' (K = 0.61–0.8) or 'almost perfect' agreement (K = 0.81–1.0), authors received further training before carrying out further searches. Authors of other synopses who have searched journals and added relevant publications to the Conservation Evidence literature database since 2018, and all other searchers since 2017 have undertaken the initial paper inclusion test described above; searchers prior to that have not. Kappa tests of the first two years searched have been carried out for all new searchers who have contributed to the Conservation Evidence literature database since July 2018.

We acknowledge that the literature search and screening method used by Conservation Evidence, as with any method, will result in gaps in the evidence. The Conservation Evidence literature database currently includes relevant papers from over 300 English language journals. Additional journals are frequently added to those searched, and years searched are often updated. It is possible that searchers will have missed relevant papers from those journals searched. Publication bias, where studies reporting negative or non-significant findings are less likely to be written up and published in journals (e.g. Dwan *et al.* 2013), was not taken into account, and it is likely that additional biases will result from the evidence that is available, for example geographic biases in study locations.

b) Inclusion criteria

The following Conservation Evidence inclusion criteria were used.

Criteria A: Conservation Evidence includes studies that measure the effect of an intervention that might be done to conserve biodiversity

1. Does this study measure the effect of an intervention that is or was under the control of humans, on wild taxa (including captives), habitats, or invasive/problematic taxa? If yes, go to 3. If no, go to 2.
2. Does this study measure the effect of an intervention that is or was under the control of humans, on human behaviour that is relevant to conserving biodiversity? If yes, go to Criteria B. If no, the study will be excluded.
3. Could the intervention be put in place by a conservationist/decision maker to protect, manage or restore wild taxa or habitats, reduce impacts of threats to wild taxa or habitats, or control or mitigate the impact of an invasive/problematic taxon on wild taxa or habitats? If yes, the study will be included. If no, the study will be excluded.

Explanation:

1.a. Study must have a measured outcome on wild taxa, habitats or invasive species: excludes studies on domestic/agricultural species, theoretical modelling or opinion pieces. See Criteria B for interventions that have a measured outcome on human behaviour only.

1.b. Intervention must be carried out by people: excludes impacts from natural processes (e.g. wave action, natural storms), impacts from background variation (e.g. sediment type, climate change), correlations with habitat types, where there is no test of a specific intervention by humans, or pure ecology (e.g. movement, distribution of species).

2. Study must test an intervention that could be put in place for conservation. This excludes assessing impacts of threats (interventions which remove threats would be included). The test may involve comparisons between sites/factors not originally put in place or modified for conservation, but which could be (e.g. fished vs unfished sites, dredged vs undredged sites –where the removal of fishing/dredging is as you would do for conservation, even if that was not the original intention in the study).

If the title and/or abstract are suggestive of fulfilling our criteria, but there is not sufficient information to judge whether the intervention was under human control, the intervention could be applied by a conservationist/decision maker or whether there are data quantifying the outcome, then the study will be included. If the article has no abstract, but the title is suggestive, then a study will be included.

We sort articles into folders by which taxon/habitat they have an outcome on. If the title/abstract does not specify which species/taxa/habitats are impacted, then the full article will be searched and then assigned to folders accordingly.

The outcome for wild taxa/habitats can be negative, neutral or positive, does not have to be statistically significant but must be quantified (if hard to judge from abstract, then it will be included). It could be any outcome that has implications for the health of individuals, populations, species, communities or habitats, including, but not limited to the following:

- Individual health, condition or behaviour, including in captivity: e.g., growth, size, weight, stress, disease levels or immune function, movement, use of natural/artificial habitat/structure, range, or predatory or nuisance behaviour that could lead to retaliatory action by humans
- Breeding: egg/sperm production, sperm motility/viability after freezing, artificial fertilization success, mating success, birth rate, litter size, offspring condition, 'overall recruitment'
- Genetics: genetic diversity, genetic suitability (e.g. adaptation to local conditions, use of correct routes for migratory species, etc.)
- Life history: age/size at (sexual) maturity, survival, mortality
- Population measures: number, abundance, density, presence/absence, biomass, movement, cover, age-structure, species distributions (only in response to a human action), disease prevalence, sex ratio

- Community/habitat measures: species richness, diversity measures (including trait/functional diversity), community composition, community structure (e.g. trophic structure), area covered (e.g. by different habitat types), physical habitat structure (e.g. rugosity, height, basal area)

Interventions within the scope of Conservation Evidence include:

- Clear management interventions: e.g. closing an area to fishing, modifying fishing gear to reduce bycatch, controlling invasive species, creating or restoring habitats
- International or national policies
- Reintroductions or management of wild species in captivity
- Interventions that reduce human-wildlife conflict
- Interventions that change human behaviour, resulting in an impact on wild taxa or habitats

See https://www.conservationevidence.com/data/index for more examples of interventions.

Note on study types:

Literature reviews, systematic reviews, meta-analyses or short notes that review studies that fulfil these criteria will be included.

Theoretical modelling studies will be excluded, as no intervention has been taken. However, studies that use models to analyse real-world data, or compare models to real-world situations will be included (if they otherwise fulfil these criteria).

Criteria B: Conservation Evidence includes studies that measure the effect of an intervention that might be done to change human behaviour for the benefit of biodiversity

1. Does this study measure the effect of an intervention that is or was under human control on human behaviour (actual or intentional) which is likely to protect, manage or restore wild taxa or habitats, or reduce threats to wild taxa or habitats? If yes, go to 2. If no, the study will be excluded.
2. Could the intervention be put in place by a conservationist, manager or decision maker to change human behaviour? If yes, the study will be included. If no, the study will be excluded.

Explanation:

1.a. Study must have a measured outcome on actual or intentional human behaviour including self-reported behaviours: excludes outcomes on human psychology (tolerance, knowledge, awareness, attitude, perceptions or beliefs).

1.b. Change in human behaviour must be linked to outcomes for wild taxa and habitats, excludes changes in behaviour linked to outcomes for human benefit, even if these occurred under a conservation program (e.g. we would exclude a study demonstrating increased school attendance in villages under a community based conservation program).

1.c. Intervention must be under human control: excludes impacts from climatic or other natural events.

2. Study must test an intervention that could be put in place for conservation: excludes studies with no intervention, e.g. correlating human personality traits with likelihood of conservation-related behaviours.

The human behaviour outcome of the study can be negative, neutral or positive, does not have to be statistically significant but must be quantified (if hard to judge from abstract, then it will be included). It could be any behaviour that is likely to have an outcome on wild taxa and habitats (including mitigating the impact of an invasive/problematic taxon on wild taxa or habitats).

Interventions include, but are not limited to the following:

- Change in adverse behaviours (which directly threaten biodiversity) e.g. unsustainable fishing (industrial, artisanal or recreational), urban encroachment, creating noise, entering sensitive areas, polluting or dumping waste, clearing or habitat destruction, introducing invasive species
- Change in positive behaviours e.g. uptake of alternative/sustainable livelihoods, number of households adopting sustainable practices, donations
- Change in policy or conservation methods e.g. designation of protected areas, protection of key habitats/species
- Change in consumer or market behaviour e.g. purchasing, consuming, buying, willingness to pay, selling, illegal trading, advertising, consumer fraud
- Behavioural intentions to do any of the above

Interventions which are particularly likely to have a behaviour change outcome include, but are not limited to the following:

- Enforcement: closed seasons, size limits, fishing gear/hunting restrictions, auditable/traceable reporting requirements, market inspections, increase number of rangers, patrols or frequency of patrols in, around or within protected areas, improved fencing/physical barriers, improved signage, improve equipment/technology used by guards

- Behaviour change: promote alternative/sustainable livelihoods, payment for ecosystem services, ecotourism, poverty reduction, debunking misinformation, altering or re-enforcing local taboos, financial incentives
- Governance: protect or reward whistle-blowers, increase government transparency, ensure independence of judiciary, provide legal aid
- Market regulation: trade bans, taxation, supply chain transparency laws, annual harvest/export quotas
- Consumer demand reduction: fear appeals (negative association with undesirable product), benefit appeal (positive association with desirable behaviour), worldview framing, moral framing, employing decision defaults, providing decision support tools, simplifying advice to consumers, promoting desirable social norms, legislative prohibition
- Sustainable alternatives: certification schemes, captive bred or artificial alternatives, sustainable alternatives
- New policies for conservation/protection

We allocate studies to folders by their outcome. All studies under Criteria B go in the 'Behaviour change' folder. They are additionally duplicated into a taxon/habitat folder if there is a specific intended final outcome of the behaviour change (if none mentioned, they will be filed only in Behaviour change).

c) Relevant subject

Studies relevant to the synopsis subject include those focused on the conservation of wild, native, reptiles (Crocodilia, Testudines, Squamata, Rhynchocephalia).

d) Relevant types of intervention

An intervention has to be one that could be put in place by a manager, conservationist, policy maker, advisor, consultant or scientific authority to protect, manage or restore wild, native reptiles or reduce the impacts of threats to them. Alternatively, interventions may aim to change human behaviour (actual or intentional), which is likely to protect, manage or restore wild, native reptiles or reduce threats to them. See inclusion criteria above for further details.

If the following two criteria were met, a combined intervention was created within the synopsis, rather than duplicating evidence under all the separate interventions: a) there were five or more publications that used the same well-defined combination of interventions, with a clear description of what they were, without separating the effects of each individual intervention, and b) the combined set of interventions is a commonly used conservation strategy.

e) Relevant types of comparator

To determine the effectiveness of interventions, studies must include a comparison, i.e. monitoring change over time (typically before and after the intervention was implemented), or for example at treatment and control sites. Alternatively, a study could compare one specific intervention (or implementation method) against another. For example, this could be comparing the abundance of a turtle species before and after the closure of an area to fishing activities, or the reduction in reptile bycatch using different types of fishing gear. Exceptions, which may not have a control but were still included, are for example the effectiveness of captive breeding or rehabilitation programmes.

f) Relevant types of outcome

Below we provide a list of anticipated metrics; others were included if reported within relevant studies.

- Community response
    - Community composition
    - Richness/diversity
- Population response
    - Abundance: number, density, presence/absence
    - Reproductive success: egg/sperm production, artificial fertilization success, mating success, birth rate, hatchling quality/condition, overall recruitment, age/size at maturity
    - Survival: survival rates, mortality
    - Condition: growth, size, weight, condition factors, biochemical ratios, stress, energetics, disease levels or immune function, genetic diversity
- Behaviour
    - Use of natural/artificial habitat/structure
    - Behaviour change: movement, range, timing (e.g. of migration, foraging period)
- Other
    - Reduction in entanglements/unwanted catch ('bycatch')
    - Change in human behaviour
    - Human wildlife conflict
    - Offspring sex ratio

g) Relevant types of study design

The table below lists the study designs included. The strongest evidence comes from replicated, randomized, controlled trials with paired sites and before-and-after monitoring.

Table 1. Study designs

Term	**Meaning**
Replicated	The intervention was repeated on more than one individual or site. In conservation and ecology, the number of replicates is much smaller than it would be for medical trials (when thousands of individuals are often tested). If the replicates are sites, pragmatism dictates that between five and ten replicates is a reasonable amount of replication, although more would be preferable. We provide the number of replicates wherever possible. Replicates should reflect the number of times an intervention has been independently carried out, from the perspective of the study subject. For example, 10 plots within a mown field might be independent replicates from the perspective of plants with limited dispersal, but not independent replicates for larger motile animals such as birds. In the case of translocations/release of captive bred animals, replicates should be sites, not individuals. In the case of captive-breeding programmes, studies were considered to be replicated when at least 5 breeding females were included.
Randomized	The intervention was allocated randomly to individuals or sites. This means that the initial condition of those given the intervention is less likely to bias the outcome.
Paired sites	Sites are considered in pairs, within which one was treated with the intervention and the other was not. Pairs, or blocks, of sites are selected with similar environmental conditions, such as water quality or adjacent land use. This approach aims to reduce environmental variation and make it easier to detect a true effect of the intervention.
Controlled*	Individuals or sites treated with the intervention are compared with control individuals or sites not treated with the intervention. (The treatment is usually allocated by the investigators (randomly or not), such that the treatment or control groups/sites could have received the treatment).
Before-and-after	Monitoring of effects was carried out before and after the intervention was imposed.
Site comparison*	A study that considers the effects of interventions by comparing sites that historically had different interventions (e.g. intervention vs no intervention) or levels of intervention. Unlike controlled studies, it is not clear how the interventions were allocated to sites (i.e. the investigators did not allocate the treatment to some of the sites).
Review	A conventional review of literature. Generally, these have not used an agreed search protocol or quantitative assessment of the evidence.

Systematic review	A systematic review follows structured, predefined methods to comprehensively collate and synthesise existing evidence. It must weight or evaluate studies, in some way, according to the strength of evidence they offer (e.g. sample size and rigour of design). Environmental systematic reviews are available at: www.environmentalevidence.org/index.htm
Study	If none of the above apply, for example a study measuring change over time in only one site or only after an intervention. Or a study measuring use of nest boxes at one site.

*Note that "controlled" is mutually exclusive from "site comparison". A comparison cannot be both controlled and a site comparison. However, one study might contain both controlled and site comparison aspects, e.g. study of bycatch by fishers using modified nets (e.g. with a smaller mesh size) and unmodified nets (controlled), and fishers using an alternative net modification, e.g. stiffened nets (site comparison).

Study quality assessment & critical appraisal

We did not quantitatively assess the evidence from each publication or weight it according to quality. However, to allow interpretation of the evidence, we made the size and design of each study we reported clear. We critically appraised each potentially relevant study and excluded those that did not provide data for a comparison to the treatment, did not statistically analyse the results (or if included this was stated in the summary paragraph) or had obvious errors in their design or analysis. A record of the reason for excluding any of the publications was included during screening and kept within the synopsis database.

Data extraction

Data on the effectiveness of the relevant intervention (e.g. mean species abundance inside or outside a protected area; reduction in bycatch after installation of a bycatch reduction device) was extracted from and summarized for publications that included the relevant subject, types of intervention, comparator and outcomes outlined above. A summary of the total number of evidence sources and papers/reports searched and the total number of publications included following data extraction is presented in Appendix 3.

In addition to ensuring consistency/accuracy when screening publications for inclusion in the discipline-wide literature database (see above), when authors first began summarising, the first 10 publications were sent to Conservation Evidence for editing. Further to this, relevant data were extracted by a member of the core Conservation Evidence team for a set of publications as well as the synopsis author to ensure agreement on the correct data and interpretation of the results for inclusion in the synopsis. In addition, summaries were also swapped between authors on a semi-regular basis to quality control the paragraphs that were being written.

Evidence synthesis

a) Summary protocol

Each publication usually has just one paragraph for each intervention it tests describing the study in (usually) no more than 150 words using plain English, though more complex studies required longer summaries. Each summary is in the following format:

A [TYPE OF STUDY] in [YEARS X-Y] in [HOW MANY SITES] in/of [HABITAT] in [REGION and COUNTRY] [REFERENCE] found that [INTERVENTION] [SUMMARY OF ALL KEY RESULTS] for [SPECIES/HABITAT TYPE]. [DETAILS OF KEY RESULTS, INCLUDING DATA]. In addition, [EXTRA RESULTS, IMPLEMENTATION OPTIONS, CONFLICTING RESULTS]. The [DETAILS OF EXPERIMENTAL DESIGN, INTERVENTION METHODS and KEY DETAILS OF SITE CONTEXT]. Data was collected in [DETAILS OF SAMPLING METHODS].

Type of study -use terms and order in Table 1.

Site context -for the sake of brevity, only nuances essential to the interpretation of the results are included. The reader is always encouraged to read the original source to get a full understanding of the study site (e.g. history of management, physical conditions, landscape context etc.).

For example:

A replicated, controlled study in 2004–2011, along 100 km of sandy beach in Rio de Janeiro State, Brazil (1) found that relocating loggerhead turtles *Caretta caretta* nests to nearby locations on the beach resulted in lower hatching success compared to nests left in situ. Hatching success was lower for relocated nests than for nests left in situ in six of seven seasons (relocated: 57–69%; in situ: 73–81%). In addition, hatching success was also lower for nests relocated to an on-beach hatchery in six of seven seasons (61–66%) compared to in situ nests. In the nesting seasons of 2004–2011 beaches were patrolled daily, and nests were transferred to a safe location on the beach (24–172 nests/season); moved to an on-beach hatchery (231–1,015 nests/season); or left in situ (8–316 nests/season). Those nests not taken to the hatchery were covered with a wire mesh screen. After hatchling emergence, nests were excavated to assess hatching success.

(1) Lima E.P.E., Wanderlinde J., de Almeida D.T., Lopez G. & Goldberg D.W. (2012) Nesting ecology and conservation of the loggerhead sea turtle (*Caretta caretta*) in Rio de Janeiro, Brazil. *Chelonian Conservation and Biology*, 11, 249–254.

A replicated, controlled study in 2004–2008 in pelagic waters in the south-western Atlantic Ocean in Brazil (2) found that using circle hooks reduced unwanted catch of sea turtles compared to J-hooks in a longline fishery. Unwanted catch of loggerhead *Caretta caretta* and leatherback *Dermochelys coriacea* were reduced when circle hooks were used (loggerhead: 0.8 turtles/1,000 hooks, leatherback: 0.7) compared to J-hooks (loggerhead: 1.9, leatherback: 1.6). Fewer loggerhead turtles swallowed hooks when circle hooks were used (6%) compared to J-hooks (25%). However, on average, circle hooks caught larger loggerheads (61 cm average carapace length) than J-hooks (58 cm). Catch rates of most target fish species was increased when circle hooks were used, with the exception of swordfish *Xiphius gladius* (see paper for details). Catch rates of 10° offset 18/0 circle hooks (2.8–2.2 cm gape width) were compared to traditional 9/0 0° offset J-hooks (2.9 cm gape width). Twenty-seven trips totalling 229 fishing trips were undertaken. A total of 145,828 baited hooks were tested by alternating hooks along sections of the mainline.

(2) Sales G., Giffoni B.B., Fiedler F.N., Azevedo V.G., Kotas J.E., Swimmer Y. & Bugoni L. (2010) Circle hook effectiveness for the mitigation of sea turtle bycatch and capture of target species in a Brazilian pelagic longline fishery. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 20, 428–436.

b) Terminology used to describe the evidence

Unless specifically stated otherwise, results reflect statistical tests performed on the data, i.e. we only state that there was a difference if it was supported by the statistical test used, and otherwise state that there was no difference or that outcomes were similar. If there was a good reason to report differences between treatments and controls that were not tested for statistical significance, it was made clear within the summary that statistical tests were not carried out. Table 1 above defines the terms used to describe the study designs.

c) Dealing with multiple interventions within a publication

When separate results were provided for the effects of each of the different interventions tested, separate summaries were written under each intervention heading. However, when several interventions were carried out at the same time and only the combined effect reported, the result was described with a similar paragraph under all relevant interventions. In these circumstances, we clearly communicated within the summary paragraph where multiple interventions were used in combination. For example, the first sentence would articulate that a combination of interventions were carried out, i.e. '...(REF) found that [x intervention], along with [y] and [z interventions] resulted in [describe effects]'.

d) Dealing with multiple publications reporting the same results and reviews

If two publications described results from the same intervention implemented in the same space and at the same time, we only included the most stringently peer-reviewed publication (i.e. journal of the highest impact factor). If one included initial results (e.g. after year one) of another (e.g. after 1–3 years), we only included the publication covering the longest time span. If two publications described at least partially different results, we included both but made clear they were from the same project in the paragraph, e.g. 'A controlled study... (Gallagher *et al*. 1999; same experimental set-up as Oasis *et al*. 2001)...'.

e) Taxonomy

Taxonomy was not updated but follows that used in the original publication. Where possible, common names and scientific names were both given the first time each species was mentioned within each summary.

f) Key messages

Each intervention has a set of concise, bulleted key messages at the top, written once all the literature had been summarized. These include information such as the number, design and location of studies included. The first bullet point describes the total number of studies that tested the intervention and the locations of the studies, followed by key information on the relevant metrics presented under the headings and sub-headings shown below (with number of relevant studies in parentheses for each).

- **X studies** examined the effects of [INTERVENTION] on [TARGET POPULATION]. Y studies were in [LOCATION 1][1,2] and Z studies were in [LOCATION 2][3,4].
  Locations will usually be countries, ordered based on chronological order of studies rather than alphabetically, i.e. 'the USA[1], Australia[2]' rather than 'Australia[2], the USA[1]'. However, when more than 4–5 separate countries, they may be grouped into regions to make it clearer e.g. Europe, North America. The distribution of studies amongst habitat types may also be added here if relevant.

COMMUNITY RESPONSE (x STUDIES)

- **Community composition (x studies):**
- **Richness/diversity (x studies):**

POPULATION RESPONSE (x STUDIES)

- **Abundance (x studies):**
- **Reproductive success (x studies):**
- **Survival (x studies):**
- **Condition (x studies):**

BEHAVIOUR (x STUDIES)

- **Use (x studies):**
- **Behaviour change (x studies):**

OTHER (x STUDIES) *(Included only for interventions/chapters where relevant)*

- **[Sub-heading(s) for the metric(s) reported will be created] (x studies):**

If no suitable studies are found for an intervention, the following text was added in place of the key messages above:

- We found no studies that evaluated the effects of [INTERVENTION] on [TARGET POPULATION].

  'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

g) Background information

Background information for an intervention is provided to describe the intervention and where we feel recent knowledge is required to interpret the evidence. This is presented after the key messages and relevant references are included in a reference list at the end of the Background section. In some cases, where a body of literature has strong implications for reptile conservation, but does not directly test interventions for their effects, we may also refer the reader to this literature in the background sections.

Dissemination/communication of evidence synthesis

The information from this evidence synthesis is available in three ways:

- A synopsis pdf, downloadable from www.conservationevidence.com, contains the study summaries, key messages and background information on each intervention.
- The searchable database at www.conservationevidence.com contains all the summarized information from the synopsis, along with expert assessment scores.
- A chapter in What Works in Conservation, available as a pdf to download and a book from www.conservationevidence.com/content/page/79, contains the key messages from the synopsis as well as expert assessment scores on the effectiveness and certainty of the synopsis, with links to the online database.

How you can help to change conservation practice

If you know of evidence relating to reptile conservation that is not included in this synopsis, we invite you to contact us via our website www.conservationevidence.com. If you have new, unpublished evidence, you can submit a paper to the Conservation

Evidence journal (https://conservationevidencejournal.com/). We particularly welcome papers submitted by conservation practitioners.

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Fordham D., Georges A., Corey B. & Brook B.W. (2006) Feral pig predation threatens the indigenous harvest and local persistence of snake-necked turtles in northern Australia. *Biological Conservation*, 133, 379–388.

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Pincheira-Donoso D., Bauer A.M., Meiri S. & Uetz P. (2013) Global taxonomic diversity of living reptiles. *PloS one*, 8, e59741.

Seigel R.A., Smith R.B. & Seigel N.A. (2003) Swine flu or 1918 pandemic? Upper respiratory tract disease and the sudden mortality of gopher tortoises (*Gopherus polyphemus*) on a protected habitat in Florida. *Journal of Herpetology*, 137–144.

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Van Cao N., Tao N.T., Moore A., Montoya A., Rasmussen A.R., Broad K., Voris H.K. & Takacs Z. (2014) Sea snake harvest in the Gulf of Thailand. *Conservation Biology*, 28, 1677–1687.

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2. Threat: Residential and commercial development

Background

The three greatest threats to reptile persistence from development are related to direct threats of habitat destruction and/or fragmentation, and indirect threats of associated pollution and impacts of transportation and service corridors (Gibbons *et al.* 2000). Interventions in response to these threats are described in the following chapters: *Habitat protection, Habitat restoration and creation, Threat: Pollution and Threat: Transportation and service corridors*.

The interventions that are more specific to development, including development of recreational facilities, are discussed in this section.

Residential development can result in an increase in populations of domestic cats *Felis catus* and dogs *Canis lupus familiaris,* which can prey on wildlife including reptiles. For interventions that aim to reduce predation by cats and dogs in residential areas, see *Threat: Invasive alien and other problematic species - Use collar-mounted devices to reduce predation by domestic animals*; *Keep domestic cats indoors at times when reptiles are most active* and *Leash or restrict domestic dog movements in reptile habitats.*

For studies that examine the effects of translocating reptiles away from threats, including development activity, see S*pecies Management – Mitigation translocations.*

Gibbons J.W., Scott D.E., Ryan T.J., Buhlmann K.A., Tuberville T.D., Metts B.S., Greene J.L., Mills T., Leiden Y., Poppy S. & Winne, C.T. (2000) The global decline of reptiles, *déjà vu* amphibians. *BioScience*, 50, 653–666.

2.1. Protect greenfield sites or undeveloped land in urban areas

- We found no studies that evaluated the effects of protecting greenfield sites or undeveloped land in urban areas on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

'Greenfield sites' are areas of previously undeveloped land within urban areas, such as agricultural and amenity land, forests, parks and gardens. Such sites may provide important habitat for wildlife and act as wildlife corridors. However, greenfield sites are frequently built upon with the growing pressure for urban development.

2.2. Protect brownfield or ex-industrial sites in urban areas

- **One study** evaluated the effects of protecting brownfield or ex-industrial sites in urban areas. This study was in the UK[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One study in the UK[1] reported that an ex-industrial site that was protected was occupied by up to four species of reptiles.

Background

Brownfield sites include land that was once used for industrial or other human activity but is then left disused or partially used, for example disused quarries or mines, demolished or derelict factory sites, derelict farm buildings, derelict railways or contaminated land. Natural recolonization of these sites can result in valuable habitats for wildlife and provide migration corridors in built-up or disturbed areas.

A study in 2005 in an area of mixed ponds, grassland and scrub in Peterborough, UK (1) found that following protection of an ex-industrial site, the area was occupied by grass snakes *Natrix helvetica* and common lizards *Zootoca vivipara*. A total of 87 grass snakes and 76 common lizards were recorded at the site. Authors reported that adders *Vipera berus* and slow worms *Anguis fragilis* were also present at the site (no data provided). In the 1940s to the late 1990s, the area was used for clay extraction for brick making, resulting in a landscape characterised by a series of ridges and furrows. In 1995, part of the site was designated as a Site of Special Scientific Interest, and in 2005 it was designated as a Special Area of Conservation. Authors reported that a range of habitat and species management activities were carried out on the site, including controlling scrub, construction of artificial refuges and releases of grass snakes and common lizards. In 2005, reptile surveys were conducted by placing 90 corrugated iron refugia (1 m^2) throughout vegetated locations on the site. Refugia were visited 22 times (roughly weekly visits) and all reptiles were counted.

(1) Langton T. (2006) *Western periphery road stages 2 & 3, Hampton, Peterborough*. Herpetofauna Consultants International Ltd.

2.3. Plant native species for reptile habitat in urban areas

- We found no studies that evaluated the effects of planting native species for reptile habitat in urban areas on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Planting native species is commonly used in urban settings as a means to improve biodiversity (McMahan 2006) and to improve habitat for wildlife. For studies focused on other methods of improving habitat, see *Habitat restoration and creation*.

McMahan L.R. (2006) Understanding cultural reasons for the increase in both restoration efforts and gardening with native plants. *Native Plants Journal*, 7, 31–34.

2.4. Create suitable habitats to offset habitat lost within development footprint

- We found no studies that evaluated the effects of creating suitable habitats to offset habitat lost within a development footprint on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Development activity may be accompanied by a biodiversity 'offsetting' strategy that aims to compensate for the loss of existing habitat and associated species by protecting other sites or creating new sites of equal value to the lost habitat (e.g. Bull *et al*. 2014, Ives & Bekessy 2015). Offsetting may be 'in-kind', whereby new habitat is similar to that lost (for example creating ponds to replace ponds lost elsewhere), or 'out-of-kind', whereby the new habitat is different (for example creating rock outcrops to replace lost grasslands elsewhere). Existing green spaces such as golf courses — known to support substantial wildlife numbers, especially urban-adapted species (Hodgkison *et al*. 2007) — have been proposed as potential sites for biodiversity offsets (Burgin & Wotherspoon 2009). Studies describing habitat restoration that is not compensatory for urban development or is carried out retrospectively rather than planned alongside the development, are summarized under *Habitat restoration and creation*.

Bull J.W., Gordon A., Law E.A., Suttle K.B. & Milner-Gulland E.J. (2014) Importance of baseline specification in evaluating conservation interventions and achieving no net loss of biodiversity. *Conservation Biology*, 28, 799–809.

Burgin S. & Wotherspoon D. (2009) The potential for golf courses to support restoration of biodiversity for BioBanking offsets. *Urban Ecosystems,* 12, 145–155.

Hodgkison S.C., Hero J.-M. & Warnken J. (2007) The conservation value of suburban golf courses in a rapidly urbanising region of Australia. *Landscape and Urban Planning*, 79, 323–337.

Ives C.D. & Bekessy S.A. (2015) The ethics of offsetting nature. *Frontiers in Ecology and the Environment*, 13, 568–573.

Maron M., Hobbs R.J., Moilanen A., Matthews J.W., Christie K., Gardner T.A., Keith D.A., Lindenmayer D.B. & McAlpine C.A. (2012) Faustian bargains? Restoration realities in the context of biodiversity offset policies. *Biological Conservation*, 155, 141–148.

Suding K.N. (2011) Toward an era of restoration in ecology: successes, failures, and opportunities ahead. *Annual Review of Ecology, Evolution, and Systematics*, 42, 465–487.

2.5. Erect fencing to exclude reptiles from construction zones

- We found no studies that evaluated the effects on reptile populations of erecting fencing to exclude reptiles from construction zones.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Adding temporary fencing around construction sites may help to prevent reptiles from entering (or re-entering where individuals have been moved out of the site) such sites. Fencing might also border or surround important reptile habitat and habitat features (e.g. wetlands, nesting sites, talus slopes, areas of woody debris) adjacent to or within construction sites, thereby providing protection during construction periods.

2.6. Avoid carrying out construction work during sensitive periods

- We found no studies that evaluated the effects of avoiding carrying out construction work during sensitive periods on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Ceasing or delaying construction activity during breeding, nesting or seasonal migration periods in areas used by reptiles may reduce the impacts of development activities.

2.7. Remove invasive plant species to improve habitat within development footprints

- We found no studies that evaluated the effects on reptile populations of removing invasive plant species to improve habitat within development footprints.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Greenfield, brownfield and urban areas are often invaded by weedy species. These potentially valuable habitat areas are subject to disturbances such as introduced exotic species (both plants and animals), fire and accumulation of rubbish. For studies relating to management of invasive plant species, see *Threat: Invasive alien and other problematic species* and *Habitat restoration and creation*.

2.8. Provide training for construction workers on the potential risks to reptiles and how to mitigate disturbance during works

- We found no studies that evaluated the effects on reptile populations of providing training for construction workers on the potential risks to reptiles and how to mitigate disturbance during works.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Training construction workers to recognise potential risks to wildlife during the construction process (e.g. which areas may be important habitat within and adjacent to construction sites) and best practices to mitigate disturbance during construction (e.g. minimising compacting or disturbing the ground within the construction area) or enhance habitat during construction (e.g. taking advantage of surplus woody debris, rocks, gravel and displaced soil to optimise available habitat) may reduce the impacts of development activities (Ovaska *et al*. 2014).

Ovaska K., Sopuck L., Engelstoft C., Matthias L., Wind E. & MacGarvie J. (2014) *Guidelines for Amphibian and Reptile Conservation during Urban and Rural Land Development in British Columbia*. B.C. Government.

3. Threat: Agriculture and Aquaculture

Background

In many parts of the world, much of the conservation effort is directed at reducing the impacts of agricultural intensification on biodiversity on farmland and in the wider countryside. This chapter covers those interventions that seek to reduce the impact of both agriculture and aquaculture, and interventions are organised under three sections: terrestrial habitat management; aquatic habitat management; and marine and freshwater aquaculture. Further substantial threats from agriculture include loss of habitat and pollution (e.g. from fertilizer and pesticide use). Interventions in response to these threats are described in the following chapters: *Habitat restoration and creation*, *Threat: Natural system modifications* and *Threat: Pollution*.

3.1. Engage landowners and volunteers to manage land for reptiles

- We found no studies that evaluated the effects on reptile populations of engaging landowners and volunteers to manage land for reptiles.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Only around 15% of land and 4% of oceans are protected worldwide (UNEP WCMC & IUCN 2016), which means that it is vital to engage effectively with landowners so that they manage their land in ways that help to maintain reptile populations. Volunteers can also make a valuable contribution to the management of habitats for reptiles, on private and public land. In some cases, the long-term success of habitat management can depend on the involvement of local people.

As well as the direct effects from habitat restoration, volunteer programmes help raise awareness about reptiles and the threats that they face. For example, a study found that participants with high levels of engagement in conservation projects learned more (Evely *et al.* 2011). For interventions that involve engaging volunteers to help manage or monitor reptile populations see the chapter on *Education and awareness raising*.

UNEP-WCMC (United Nations Environment - World Conservation Monitoring Centre) & IUCN (International Union for the Conservation of Nature) (2016) *Protected Planet Report 2016*. UNEP-WCMC, Cambridge, United Kingdom.

Evely A.C., Pinard M., Reed M.S. & Fazey L. (2011) High levels of participation in conservation projects enhance learning. *Conservation Letters*, 4, 116–126.

3.2. Pay farmers to cover the costs of conservation measures

- **One study** evaluated the effects of paying farmers to cover the costs of conservation measures on reptiles. This study was in Australia[1].

COMMUNITY RESPONSE (1 STUDY)

- **Richness/diversity (1 study):** One replicated, site comparison study in Australia[1] found that sites managed under agri-environment schemes had similar reptile species richness compared to sites that were managed purely for livestock production or areas of unmanaged woodland.

POPULATION RESPONSE (1 STUDY)

- **Abundance (1 study):** One replicated, site comparison study in Australia[1] found that sites managed under agri-environment schemes had similar reptile abundances compared to sites that were managed purely for livestock production or areas of unmanaged woodland.

BEHAVIOUR (0 STUDIES)

Background

Financial incentives to undertake specific management actions with the aim of increasing biodiversity on farmland may be provided by government or non-governmental organisations. For example, agri-environment schemes are government or inter-governmental schemes designed to compensate farmers financially for changing agricultural practices to be more favourable to biodiversity and the landscape. Agri-environment schemes include many different specific interventions relevant to conservation. Where a study describes a specific intervention, e.g. *Create uncultivated margins around arable or pasture fields*, it is summarized under that specific action. Here we include studies that describe the effectiveness of payments such as those included in agri-environment policies where specific actions are not clearly defined.

A replicated, site comparison in 2007–2010 in farmed temperate woodlands in New South Wales, Australia (1) found that agri-environment schemes did not increase reptile species richness or abundance after one–three or six–eight years of conservation management compared to areas managed purely for livestock production and areas of unmanaged woodland. Overall reptile species richness and abundance was similar in sites with one–three years of agri-environment scheme management (2–3 species/site, 11–19 individuals/site) and six–eight years of agri-environment scheme management (2–4, 13–23). Sites with agri-environment schemes were also similar compared to sites managed purely for livestock production (3–4, 12–20) and sites of unmanaged woodland (2–3, 18–29). See paper for details of individual species abundances. In 2007, one hundred and five >2 ha woodland sites (of four different vegetation types) on 53 farms were established, which had been managed in one of four ways: short-term agri-environment schemes (removing or reducing livestock grazing, revegetation and control of introduced plants and animals since 2007; 16 sites); long-term agri-environment schemes (managed for biodiversity outcomes since before 2003; 32 sites); managed purely for livestock production (grazed with higher stocking densities and occasional fertilizer application; 40 sites), or unmanaged woodland (woodlands established 150 years prior, vegetation not cleared and rarely grazed, 17 sites). During October 2008, August 2009 and August 2010, reptiles were monitored in each site using 30-minute active searches under artificial refuges (four 1.2 m railway sleepers, four roof tiles and 1 m^2 pile of corrugated steel) along one 200 x 50 m transect/site.

(1) Michael D.R., Wood J.T., Crane M., Montague-Drake R. & Lindenmayer D.B. (2014) How effective are agri-environment schemes for protecting and improving herpetofaunal diversity in Australian endangered woodland ecosystems? *Journal of Applied Ecology*, 51, 494–504.

Terrestrial habitat management

3.3. Manage tillage practices

- We found no studies that evaluated the effects of managing tillage practices on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Conventional ploughing or tilling disturbs the soil to about 25–35 cm (Tebrügge & Düring 1999). Ploughing can impact reptiles when individuals are buried by ploughing activity (Saumure *et al.* 2007), or when ploughing destroys nesting sites (Hódar *et al.* 2000). A number of methods can be used to reduce the depth or intensity of ploughing such as layered cultivation, non-inversion tillage and conservation tillage. Such practices have been found to be beneficial for some farmland biodiversity (Holland & Luff 2000). 'Conservation agriculture' attempts to alter the soil profile as little as possible by direct sowing and/or leaving the soil protected with plant residues (García-Torres *et al.* 2002).

García-Torres L., Martínez-Vilela A., Holgado-Cabrera A. & Gónzalez-Sánchez E. (2002) *Conservation agriculture, environmental and economic benefits*. Summary of the Workshop on Soil Protection and Sustainable Agriculture, Soria, Spain, 15–17 May 2002, European Conservation Agriculture Federation.

Hódar J.A., Pleguezuelos J.M. & Poveda J.C. (2000) Habitat selection of the common chameleon (*Chamaeleo chamaeleon*) (L.) in an area under development in southern Spain: implications for conservation. *Biological Conservation*, 94, 63–68.

Holland J.M. & Luff M.L. (2000) The effects of agricultural practices on Carabidae in temperate agroecosystems. *Integrated Pest Management Reviews*, 5, 109–129.

Saumure R.A., Herman T.B. & Titman R.D. (2007) Effects of haying and agricultural practices on a declining species: The North American wood turtle *Glyptemys insculpta*. *Biological Conservation*, 135, 565–575.

Tebrügge F. & Düring R.-A. (1999) Reducing tillage intensity—a review of results from a long-term study in Germany. *Soil and Tillage Research*, 53, 15–28.

3.4. Manage crop diversity

- We found no studies that evaluated the effects of managing crop diversity on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Some heterogeneity in farmland is thought to be key in determining on-farm biodiversity (Benton *et al.* 2003). Therefore, increasing the range of different crops grown in a given year may increase the biological value of a farm.

Benton T.G., Vickery J.A. & Wilson J.D. (2003) Farmland biodiversity: is habitat heterogeneity the key? *Trends in Ecology & Evolution*, 18, 182–188.

3.5. Modify grazing regime

Background

Grazing by livestock changes habitats by reducing vegetation height and ground cover, altering plant abundance and diversity, creating openings for seed growth and preventing reed or shrub growth. While heavy grazing by some wild grazers can have detrimental effects on reptile populations (Howland *et al.* 2014), the result of different grazing regimes on reptiles will likely depend on the reptile species, grazing intensity and the timing of grazing activity. Studies included in this intervention measure the impacts of varying intensities of grazing or different types of grazing regimes on reptiles. Studies that just compare the effect of stopping all grazing to continued grazing are included under the intervention *Cease livestock grazing*.

Due to the number of studies found, this action has been split by habitat type.

For interventions that aim to reduce the detrimental effects of grazing by wild herbivores see *Threat: Invasive or problematic species - Remove or control invasive or problem herbivores and seed eaters*.

Howland B., Stojanovic D., Gordon I.J., Manning A.D., Fletcher D. & Lindenmayer D.B. (2014) Eaten out of house and home: impacts of grazing on ground-dwelling reptiles in Australian grasslands and grassy woodlands. *PLoS One*, 9, e105966.

Grassland & shrubland

- Four studies evaluated the effects of modifying grazing regimes in grassland and shrubland on reptile populations. Three studies were in the USA[1,2,4] and one was in Australia[3].

COMMUNITY RESPONSE (3 STUDIES)

- **Richness/diversity (3 studies):** One replicated site comparison study in the USA[1] found that sites with different grazing intensities had similar reptile diversity. One replicated, site-comparison, paired sites study in Australia[3] found no clear effects of modifying grazing intensities on reptile species richness. One replicated, controlled, before-and-after study in the USA[4] found that areas that were lightly grazed or unmanaged had lower reptile species richness than areas that were heavily grazed in combination with burning.

POPULATION RESPONSE (3 STUDIES)

- **Abundance (2 studies):** One of two replicated studies (including one site comparison, paired sites study) in the USA[1] and Australia[3] found that plots with lighter grazing had higher lizard abundance than those with heavier grazing in four of five vegetation types[1].

The other study[3] found that the abundance of individual reptile species or species groups remained similar at different grazing intensities.

- **Survival (1 study):** One site comparison study in the USA[2] found that survival of Texas horned lizards was higher in moderately grazed than heavily grazed sites.

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One replicated, controlled, before-and-after study in the USA[4] found that light grazing or heavy grazing and burning had mixed effects on the reptile species that used those areas.

A replicated, site comparison study in 1978–1979 in grass and scrubland in western Arizona, USA (1) found that overall lizard abundance but not diversity was higher under lighter grazing regimes in four of five vegetation types compared to heavier grazing. Lighter grazed plots had higher abundances of lizards compared to heavier grazing in chaparral (light: 1.7 individuals/trap group/night; heavy: 1.2), desert grassland (light: 0.8; heavy: 0.6), mixed riparian scrub (light: 1.2; heavy: 0.7) and cottonwood-willow (light: 1.1; heavy: 0.6). Relative abundances were similar in Sonoran desertscrub regardless of grazing regime (light: 1.0; heavy: 1.1). Species diversity was statistically similar between lightly and heavily grazed sites across all vegetation sites (reported as Shannon-Weaver diversity index). See paper for details of results for individual species. Seven lightly grazed and seven heavily grazed plots were established in five different vegetation communities: chaparral, desert grassland, mixed riparian scrub, riparian cottonwood-willow and Sonoran desertscrub (70 total plots). Lightly grazed sites were characterised by a lack of livestock and good habitat condition. Heavily grazed sites were characterised by existence of cattle trails, presence of livestock and poor habitat condition. Abundance and diversity were estimated using drift fences with four pitfall traps in March–June and September–November 1978 and March–October 1979.

A site comparison study in 1998–2001 in an area of thornscrub in southern Texas, USA (2) found Texas horned lizard *Phrynosoma cornutum* survival was higher under moderate grazing than heavy grazing, but highest in ungrazed sites. Survival of Texas horned lizards over four-months periods was higher in moderately-grazed sites (54%) than in heavily-grazed sites (33%) but lower than in ungrazed sites (77%). Lizard survival was monitored in a wildlife management area (6,500 ha) in three sites (50–60 ha), each with a different grazing regime: the ungrazed site had not been grazed since 1976, the moderately-grazed site was stocked at 30–50 steers/ha/day and the heavily-grazed site had 75–100 steers/ha/day. Lizards were captured by searching roads, chance encounters and drift fences with pitfall traps. Lizards were marked with a PIT tag and toe clips and fitted with a radio transmitter (ungrazed: 20 lizards, moderately grazed: 43 lizards, heavily grazed: 44 lizards). Lizards were located at least once every 24 hours for four months from mid-April to mid-August in 1998–2001.

A replicated, paired sites, site comparison study in 1993–1996 and 2007 in chenopod scrubland in South Australia, Australia (3) found overall reptile species richness and abundances did not show a clear response to different grazing intensities. Overall reptile species richness was 9 species/site in light and medium grazing sites and 10 species/site in heavy grazing sites (numbers taken from figure 6). Of 38 species recorded, no individual species or species group (agamid lizards, skinks, geckos) abundances changed in response to different grazing intensities alone (results reported as model outputs, see paper for other factors affecting individual abundances). However, gecko capture rates may have been lower in light grazing sites (8 individuals/site) compared to medium grazing (13 individuals/site), but similar to heavy grazing (8 individuals/site; number taken from figure 6). Four paired sites of differing grazing pressure were set out in 1993 (low intensity grazing: <12 cattle dung/ha; medium: 12–100; high: >120). Reptiles were sampled for 10 days in summer from 1993–1996 and again in 2007 using 300 mm long flymesh drift fences with 13 unbaited pitfall traps (500 mm deep x 150 mm wide, 8 m apart).

A replicated, controlled, before-and-after study in 2011–2012 in four riparian grasslands in Missouri, USA (4) found that light grazing resulted in lower reptile species richness compared to heavy grazing after prescribed burning, but similar richness compared to ungrazed areas. The effects of heavy grazing and burning cannot be separated and all results reported as statistical model outputs. Reptile species richness was slightly lower in lightly grazed plots and ungrazed plots compared to heavily grazed and burned plots. Turtle presence was associated with taller grass heights linked with light grazing, lizards were associated with burned and heavily grazed plots, and snakes were associated with 70–100% grass cover habitat that occurred the year following burning. Patches of four watersheds (10–54 ha) were treated with light grazing (May–July 2011 or 2012), burning followed by heavy grazing (May–July after April burning in 2011 or 2012), or unmanaged during the preceding five years. Reptile monitoring took place 2–3 times/month in March–May 2011–2012 using coverboards and visual encounter surveys.

(1) Jones K.B. (1981) Effects of grazing on lizard abundance and diversity in Western Arizona. *The Southwestern Naturalist*, 26, 107–15.
(2) Hellgren E.C. Burrow A.L., Kazmaier R.T. & Ruthven III D.C. (2010) The effects of winter burning and grazing on resources and survival of Texas horned lizards in a thornscrub ecosystem. *Journal of Wildlife Management*, 74, 300–309.
(3) Read J.L. & Cunningham R. (2010) Relative impacts of cattle grazing and feral animals on an Australian arid zone reptile and small mammal assemblage. *Austral Ecology*, 35, 314–324.
(4) Larson D. (2014) Grassland fire and cattle grazing regulate reptile and amphibian assembly among patches. *Environmental Management*, 54, 1434–1444.

Forest, open woodland & savanna

- **Seven studies** evaluated the effects of managing grazing regimes in forest, open woodland and savanna on reptile populations. Six studies were in Australia[2-7] and one was in the USA[1].

COMMUNITY RESPONSE (3 STUDIES)

- **Richness/diversity (3 studies):** One replicated site comparison study in the USA[1] found that sites with different grazing intensities had similar reptile diversity. One replicated, paired, site comparison study in Australia[5] found that farms with rotational grazing did not have higher reptile species richness than farms with continuous grazing. One replicated, site comparison study in Australia[6] found that following replanting of native vegetation, ungrazed or occasionally grazed plots had higher reptile species richness than plots that were continuously grazed.

POPULATION RESPONSE (6 STUDIES)

- **Abundance (5 studies):** One of three replicated studies (including one randomized, before-and-after study) in the USA[1] and Australia[3,7] found that areas with lighter grazing had higher lizard abundance than those with heavier grazing[1]. The other two studies[3,7] found that different grazing regimes had mixed effects on the abundance of lizards[3] and four-clawed geckos and inland snake-eyed skinks[7]. Two paired, site comparison studies (including one replicated study) in Australia[2,5] found that sites with rotational grazing had similar reptile abundance as sites with continuous grazing.
- **Occupancy/range (1 study):** One replicated, site comparison study in Australia[4] found that different grazing regimes had mixed effects on local colonization and extinction events of six lizard species.

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One replicated, paired, site comparison study in Australia[5] found that jacky dragons were found in sheep-grazed paddocks more frequently than in cattle-grazed paddocks.

A replicated, site comparison study in 1978–1979 in broadleaf forest in western Arizona, USA (1) found overall lizard abundance but not diversity was higher under lighter compared to heavier grazing regimes. Lighter grazed plots had higher abundances of lizards compared to heavier grazing in cottonwood-willow (Light grazing: 1.1 individuals/trap group/night; heavy grazing: 0.6). Species diversity was statistically similar between lightly and heavily grazed sites across all vegetation sites (result presented as diversity index). See paper for details of individual species abundances. Seven lightly grazed and seven heavily grazed plots were established in areas of cottonwood-willow. Lightly grazed sites were characterised by a lack of livestock and good habitat condition. Heavily grazed sites were characterised by cattle trails, presence of livestock and poor habitat condition. Abundance and diversity (Shannon-Wiener Index) were estimated using drift fences with four pitfall traps in March–June and September–November 1978 and March–October 1979.

A paired, site comparison study in 2006–2007 in grassy woodland and agricultural land in south eastern Australia (2) found that rotational grazing did not increase reptile abundance compared to continuous grazing. Reptile abundance was similar in rotationally grazed plots (2.0 reptiles/ha) compared to continuously grazed plots (1.7 reptiles/ha), but greater in grazed plot with trees (3.6 reptiles/ha) than in grazed native pasture plots (1.4 reptiles/ha) regardless of grazing system. Twelve pairs of farms of with either rotational or continuous grazing (cattle or sheep) on native pastures were selected. Rotational grazing systems (four or more paddocks grazed for <56 days at a time followed by at least 21 days of rest with more rest time than grazing time) had operated for at least five years. Paddocks on continuous grazing farms were stocked for >6 months a year. Reptiles were surveyed in two 1 ha plots/farm (one in treed and one in cleared pastureland, 48 plots in total) using coverboards and active searches in December 2006, March 2007 and October 2007.

A replicated, randomized, before-and-after study in 1999–2004 in open eucalyptus savanna in north-eastern Queensland, Australia (3) found that different cattle stocking regimes had mixed effects on reptile abundance depending on the species. All results presented as model outputs. At medium stocking rates, dubious gecko *Gehyra dubia* and shaded-litter rainbow skink *Carlia munda* abundances increased over time. At high cattle stocking rates, terrestrial gecko *Diplodactylus conspicillatus* and north-eastern firetail skink *Morethia taeniopleura* abundances decreased over time. At medium and high stocking rates, Binoe's prickly gecko *Heteronotia binoei* abundance increased, but decreased in variable/rotational stocking over time. Some species' abundances varied depending on vegetation type (see paper for details). Sixteen 1 ha plots were established (>500 m apart) in a commercial livestock station (1,041 ha). Plots were grazed at moderate stocking (4 plots), heavy stocking (4 plots), or rotational/variable stocking rates (8 plots, see paper for details). Ground cover was either mainly silverleaf ironbark *Eucalyptus melanophloia* (8 plots) or reid river box *Eucalyptus brownii* (8 plots). Reptiles were surveyed in November–April and May–October in 1999–2000 and 2003–2004 using drift fences with pitfall traps and visual encounter surveys. All plots were prescribed burned in 1999 and a second fire took place in the ironbark-dominated rotational/variable stocking plots in November 2001.

A replicated, site comparison study in 2011–2013 of 29 farms in south-eastern Australia (4) found that different grazing treatments had varying effects on the colonisation and extinction probabilities of three of six lizard species. Results are reported as statistical model outputs. Two lizard species (Boulenger's snake-eyed skink *Morethia boulengeri* and southern rainbow-skink *Carlia tetradactyla*) were more likely to colonize patches with modified low or high rotational grazing than prolonged high rotational or continuous grazing. The opposite was true for straight-browed ctenotus *Ctenotus spaldingi*, which was more likely to become extinct in patches with modified high and low rotational grazing. Colonisation and extinction probabilities for three other lizard species (ragged snake-eye skink *Cryptoblepharus pannosus*, Victoria three-toed earless skink *Hemiergis talbingoensis*, marbled geckos *Christinus marmoratus*) were not significantly affected by the grazing treatments. A total of 97 sites were surveyed on 29 farms (2–4 sites/farm with different grazing treatments) within a grazing-dominated landscape. Each site used one of four grazing treatments: modified low rotational grazing (<5 years of long-duration rotational grazing following previous continuous grazing); modified high rotational grazing (<5 years of high intensity short-duration grazing following previous continuous grazing); prolonged high rotational grazing (high-intensity short-duration grazing for >10 years); continuous grazing for >10 years. Grazing was mainly by sheep *Ovis aries* and cattle *Bos taurus*. Searches were carried out for reptiles in natural habitat and artificial refuges in two plots (0.4 ha) within each site in September 2011, 2012 and 2013.

A replicated, paired, site comparison study in 2014–2015 in 12 pastures adjacent to open grassy woodland in New South Wales, Australia (5) found that rotational grazing did not increase reptile abundance or species richness compared to continuous grazing. Over one year, farms with rotational grazing did not have higher reptile abundance or species richness than farms with continuous grazing (data not provided). One lizard species, *Amphibolurus muricatus* (common name Jacky dragon not given in study) was more likely to be present in sheep-grazed rather than cattle-grazed paddocks (results presented as statistical model outputs, see paper for details). Reptiles caught were mostly skinks (Scincidae spp.). In January 2014–March 2015, reptiles were surveyed in 12 farms grazed by sheep *Ovis aries* or cattle *Bos taurus* in paddocks directly adjacent to remnants of native open grassy woodland. Five farms had a rotational grazing regime (livestock moved between paddocks every few days and not returning to the same place for weeks or months), and seven had a continuous grazing regime (livestock left in same paddock for extended periods). Surveys were carried out using drift fences, pitfall traps and funnel traps set at 20, 50 and 80 m intervals along 180 m transects that extended from the native woodland into the grazing pasture. Surveys took place for 5 days at a time in austral spring–summer.

A replicated, site comparison study in 2013 in restored eucalypt woodland on 25 farms in New South Wales, Australia (6) found that in replanted native vegetation, areas with occasional livestock grazing or no grazing had higher reptile species richness than areas with continuous grazing. Results all reported as model outputs. The authors reported that reptile species richness was higher where the amount of leaf litter was greater and that leaf litter was reduced in plots that were continuously grazed. Fifteen reptile species were recorded. In austral spring 2013, sixty-one plots of replanted native vegetation on 25 farms were surveyed in a 150 x 120 km agricultural area in the South Western Slopes (time since replanting: 6–61 years). Ten plots each were either occasionally grazed or continuously grazed by cattle *Bos taurus* or sheep *Ovies aries* (20 plots total) and a further 41 plots were never grazed. Reptiles were surveyed in each plot using 20 minute active searches and groups of artificial refuges (corrugated steel, railway sleepers and concrete roof tiles, two groups/plot).

A replicated, site comparison study in 2015 in eucalyptus woodland in Queensland, Australia (7) found that decreasing cattle grazing intensity decreased dubious four-clawed geckos *Gehyra dubia* abundance but did not change inland snake-eyed skink *Cryptoblepharus australis* abundance. Four-clawed gecko abundance was generally lower at lower grazing intensity compared to higher grazing intensity (moderate stocking: 5 geckos/plot; rotational stocking regime: 6 geckos/plot; variable stocking: 12 geckos/plot; heavy stocking: 10 geckos/plot). Inland snake-eyed skink abundance was similar at all grazing intensities (moderate stocking: 3 lizards/plot; rotational stocking: 4 lizards/plot; variable stocking: 5 lizards/plot; heavy stocking regime: 5 lizards/plot;). Data was collected in eight 100 ha paddocks each with one of four grazing regimes (two replicates of each). The grazing regimes increased in intensity from moderate to rotational to variable to heavy stocking rates (see original paper for details). Each paddock contained three sampling sites. Lizards were monitored during seven days in February 2015 using arboreal coverboards and spotlighting. Faecal samples were collected from lizards captured by hand.

(1) Jones K.B. (1981) Effects of grazing on lizard abundance and diversity in Western Arizona. *The Southwestern Naturalist*, 26, 107–15.

(2) Dorrough J., McIntyre S., Brown G., Stol J., Barrett G., & Brown A. (2012) Differential responses of plants, reptiles and birds to grazing management, fertilizer and tree clearing. *Austral Ecology*, 37, 569–582.

(3) Kutt A.S., Vanderduys E.P. & O'Reagain P. (2012) Spatial and temporal effects of grazing management and rainfall on the vertebrate fauna of a tropical savanna. *Rangeland Journal*, 34, 173–182.
(4) Kay G.M., Mortelliti A., Tulloch A., Barton P., Florance D., Cunningham S.A. & Lindenmayer D.B. (2017) Effects of past and present livestock grazing on herpetofauna in a landscape-scale experiment. *Conservation Biology*, 31, 446–458
(5) Pulsford S.A., Driscoll D.A., Barton P.S. & Lindenmayer D.B. (2017) Remnant vegetation, plantings and fences are beneficial for reptiles in agricultural landscapes. *Journal of Applied Ecology*, 54, 1710–1719.
(6) Lindenmayer D.B., Blanchard W., Crane M., Michael D. & Sato C. (2018) Biodiversity benefits of vegetation restoration are undermined by livestock grazing. *Restoration Ecology,* 26, 1157–1164.
(7) Nordberg E.J., Murray P., Alford R. & Schwarzkopf L. (2018) Abundance, diet and prey selection of arboreal lizards in a grazed tropical woodland. *Austral Ecology*, 43, 328–338.

Wetland

- **One study** evaluated the effects of managing grazing regimes in wetlands on reptile populations. This study was in France[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Abundance (1 study):** One controlled before-and-after study in France[1] found that moderate density autumn–winter grazing and autumn–spring marsh flooding resulted in higher abundance of European pond turtles than high density spring–summer grazing and winter–spring marsh flooding or low year-round grazing and flooding.
- **Condition (1 study):** One controlled before-and-after study in France[1] found that high-density spring–summer grazing resulted in fewer incidences of trampling compared to moderate-density autumn–winter grazing or low-density year-round grazing.

BEHAVIOUR (0 STUDIES)

A controlled, before-and-after study in 1997–2013 in two marshes with canals in Camargue, France (1) found that autumn–winter grazing and autumn–spring flooding increased European pond turtle *Emys orbicularis* abundance compared to high density spring–summer grazing and winter–spring marsh flooding. European pond turtle abundance was greater with moderate density autumn–winter grazing and autumn–spring marsh flooding (192–436 individuals), compared to high density spring–summer grazing and winter–spring marsh flooding (107–182 individuals) or low year-round grazing and flooding (182–227 individuals). In a nearby site with moderate year-round grazing and flooding, European pond turtle abundance was stable over the same time period (29–153 individuals). Incidences of trampling by grazing animals were higher with moderate-density autumn–winter grazing (10 individuals) or low-density year-round grazing (13 individuals) compared to high-density spring–summer grazing (4 individuals; results were not statistically tested). In 1997–2001, two sites (100–250 ha, 1.5 km apart) were flooded and grazed year-round at low-moderate stocking density. In one site, in 2002–2006, water levels were modified to create a dry period in summer–autumn, with natural flooding in winter–spring and grazing was changed to high density stocking in spring–summer (see original paper for details). In the same site, in 2007–2013, the flooding period was extended so that autumn–spring were flooded and only summer was dry, and moderate density grazing took place in autumn–winter. In April–August 1997–2013, turtles were live-trapped at both sites (7,059 total captures of 963 individuals).

(1) Ficheux S., Olivier A., Fay R., Crivelli A., Besnard A. & Bechet A. (2014) Rapid response of a long-lived species to improved water and grazing management: The case of the European pond turtle (*Emys orbicularis*) in the Camargue, France. *Journal for Nature Conservation*, 22, 342–348.

3.6. Cease livestock grazing

Background

Grazing by livestock reduces vegetation height and ground cover, alters plant abundance and diversity, creating openings for seed growth and preventing reed or shrub growth. These changes can have beneficial (Tesauro & Ehrenfeld 2007) or detrimental effects (Howland *et al.* 2014) on reptile populations depending on the reptile species, grazing intensity, timing and conjunction with burning regimes. Studies included in this intervention measure the impact of ceasing grazing on reptiles. Studies that compare the effects of varying intensities of grazing or different types of grazing regimes on reptiles are included under the intervention *Modify grazing regime*.

Due to the number of studies found, this action has been split by habitat type.

For interventions that aim to reduce the detrimental effects of grazing by wild herbivores see *Threat: Invasive or problematic species - Remove or control invasive or problem herbivores and seed eaters*.

Howland B., Stojanovic D., Gordon I.J., Manning A.D., Fletcher D. & Lindenmayer D.B. (2014) Eaten out of house and home: impacts of grazing on ground-dwelling reptiles in Australian grasslands and grassy woodlands. *PLoS One*, 9, e105966.

Tesauro J. & Ehrenfeld D. (2007) The effects of livestock grazing on the bog turtle [*Glyptemys* (=*Clemmys*) *muhlenbergii*]. *Herpetologica*, 63, 293–300.

Grassland & shrubland

- **Fifteen studies** evaluated the effects of ceasing livestock grazing in grassland and shrubland on reptile populations. Eight studies were in the USA[1,2,4-6,9,11,12], three were in Australia[7,10,15], two were in the UK[13,14] and one was in each of New Zealand[3] and Egypt[8].

COMMUNITY RESPONSE (6 STUDIES)

- **Richness/diversity (6 studies):** Four of six studies (including one replicated, controlled, before-and-after study) in the USA[1,4,6,9] and Australia[7,15] found that ungrazed and grazed areas had similar reptile species richness[7,9], combined reptile and amphibian[6] or reptile and small mammal species richness[15]. One study[1] found that ungrazed sites had higher species richness than grazed sites. The other study[4] found that fencing areas to exclude grazers had mixed effects on lizard species richness.

POPULATION RESPONSE (15 STUDIES)

- **Abundance (15 studies):** Seven of 14 studies (including one replicated, controlled, before-and-after study) in the USA[1,2,4,5,6,9,11], New Zealand[3], Australia[7,10,15], Egypt[8] and the UK[13,14] found that ceasing grazing (in one case after eradicating invasive mice[3] and in one case after burning[11]) had mixed effects on reptile[6,9,10,14] or lizard[3,4] abundance. Four studies[1,2,8,13] found that ungrazed areas had a higher abundance of lizards[1,2,8] or smooth snakes[13] than grazed areas. The other three studies[5,7,15] found that ungrazed and grazed areas had a similar abundance of reptiles[7], reptiles and small mammals[15] or Texas tortoises[5]. One replicated, randomized, site comparison study in the USA[12] found that areas with fencing that excluded both grazing and recreational vehicle use had more Agassiz's desert tortoises than areas with less restrictions on grazing or vehicle use.
- **Survival (2 studies):** One of two replicated studies (including one controlled study) in the USA[5,12] found that areas with fencing that excluded grazing and recreational vehicle use had lower death rates of Agassiz's desert tortoises than areas with less restrictions on grazing or vehicle use[12]. The other study[5] found that in areas where grazing was ceased and where grazing was rotational, survival of Texas tortoises was similar.
- **Condition (1 study):** One replicated, controlled study in the USA[5] found that in areas where grazing was ceased and where grazing was rotational, size and growth of Texas tortoises was similar.

BEHAVIOUR (1 STUDY)

- **Behaviour change (1 study):** One site comparison study in Egypt[8] found that in areas protected from grazing with fences, Be'er Sheva fringe-fingered lizards spent less time moving and were observed further away from the nearest vegetation compared to in areas with grazing and low-impact watermelon farming.

A site comparison study (year not provided) in the Mojave Desert California, USA (1) found that an ungrazed site had twice the number of lizards and two more species compared to a grazed site. Results were not statistically tested. In total, 36 lizards from five species were recorded in an ungrazed site (4 desert horned lizards *Phrynosoma platyrhinos*, 6 zebra-tailed lizards *Callisaurus draconoides*, 3 long-nosed leopard lizards *Gambelia wislizenii*, 11 common side-blotched lizards *Uta stansburiana*, 12 western whiptails *Aspidoscelis tigris*) compared to 17 lizards from three species in a grazed site (11 zebra-tailed lizards, 5 common side-blotched lizards, 1 western whiptail). Lizard surveys were carried out in May in a 100 x 100 m plot in one site with no grazing and in one site heavily grazed by sheep.

A site comparison study in 1989 in semi-arid grassland and oak savanna in Arizona, USA (2) found bunchgrass lizard *Sceloporus scalaris slevini* abundance was higher in ungrazed areas compared to grazed areas. In nine hours of searching ungrazed grassland, 41 lizards were observed compared to three lizards in nine hours of searching in grazed grassland. The ungrazed area (in a 3,160 ha ranch sanctuary) had not been grazed by livestock since 1967. The adjacent grazed area had been grazed for over a century. Abundance was determined by active searches counting the numbers of lizards over nine days in August 1989.

A before-and-after study in 1986–1993 on Mana Island, New Zealand (3) found that following removal of cattle (and cessation of grazing), and subsequent eradication of an invasive mouse *Mus musculus* the abundance of one of four lizard species decreased, two remained stable, and one increased. Before-and-after comparisons were not statistically tested. Fewer copper skinks *Cyclodina aenea* were caught after grazing stopped (1–4 captures/100 trap nights) compared to before (9 captures/100 trap nights). In the four years following mouse eradication (when grazers were still absent), the number of McGregor's skinks *Cyclodina macgregori* increased from 1 to 10 captures/100 trap nights, though numbers were similar during grazing and in the first two years after grazing stopped (6–8 captures/100 trap nights). More common geckos *Hoplodactylus maculatus* were caught when there was no grazing and mice had been eradicated (35–70 captures/100 trap nights) compared to before eradication (15 captures/100 trap nights) and during grazing (5 captures/100 trap nights). A similar number of common skinks *Leiolopisma nigriplantare polychrome* were captured after grazing ceased (after: 6–21 captures/100 trap nights) compared to before (12 captures/100 trap nights). Cattle were removed from the island in 1986–1987, and the mouse population was eradicated using poison baits in 1989–1990. In 1985–1993, lizards were trapped annually (3–8 sessions/year; 2–4 days trapping/session) using pitfall traps (582–4,066 trap nights/session) that were deployed across 27 trapping stations around the island.

A replicated, site comparison study in 1994–1996 in desert shrub and grassland in south-central California, USA (4) found that lizard abundance and species richness was higher or similar inside a protected area fenced to prevent sheep grazing, compared to grazed areas outside of the fenceline, depending on survey month and site. Lizard abundance was higher in three of six survey comparisons inside a fenced protected area without sheep grazing (4–10 lizards/transect) compared to outside of it (2–4 lizards/transect) but similar in the remaining three comparisons (inside: 2–5 lizards/transect; outside: 1–3 lizards/transect; see original paper for details). Lizard species richness was higher in one of six comparisons inside the protected area (2 species/transect) compared to outside of it (1 species/transect) but similar in the remaining five comparisons (inside: 2–3 species/transect; outside: 1–3 species/transect; see original paper for details). In 1994, two sites were selected near the north-eastern and southern boundary of the Desert Tortoise Research Natural Area (where off-road vehicles were prohibited from 1973, sheep grazing prohibited from 1978 and the boundary was fenced in 1980). Two 2.25 ha plots were established/site: one ≥400m inside the boundary and one outside the boundary (used by off-road vehicles until 1980 and grazed by sheep until 1994). In each plot, lizards were surveyed using 1.25 km transects in July 1994 and May and July 1995 (six surveys/site).

A replicated, controlled study in 1994–1997 in four pastures in chaparral shrubland in Texas, USA (5) found that excluding grazing from pastures resulted in similar abundance, survival and size of Texas tortoises *Gopherus berlandieri* compared to pastures with rotational grazing. The abundance of tortoises was similar in ungrazed (4 tortoises/100 km and 3 tortoises/10 hours) and grazed pastures (5 tortoises/100 km and 4 tortoises/10 hours). Annual survival of radio tracked individuals was similar in ungrazed (70–83%) and grazed pastures (73–84%), and size and growth were also similar (see paper for details). Two pastures each were ungrazed and grazed. Grazing was rotational (October–May) and stocking densities varied (0.2–0.6 animal units/ha; animal unit = 2 steers), though impacts on herbaceous vegetation were similar. Tortoises were counted by driving along tracks (recording the distance and time travelled) throughout the pastures between 7 April 1994 and 12 October 1997. Search effort was equal across months and time of day, and between grazed and ungrazed pastures. Forty-seven tortoises were also radiotracked.

A replicated, randomized, controlled study in 1998–1999 in streams through pasture and associated riparian areas on farms in Pennsylvania, USA (6) found that excluding livestock grazing did not increase combined reptile and amphibian species richness or abundance within 1–3 years. Overall reptile and amphibian species richness and abundance, and overall snake and turtle abundances were similar between sites fenced to exclude livestock and unfenced grazed sites (results reported as statistical tests). Of three snake species detected, abundances were higher in fenced compared to unfenced sites for northern queen *Regina septemvittata* (fenced: 6 individuals/site; unfenced: 2) and eastern garter snake *Thamnophis sirtalis* (fenced: 5; unfenced: 2) and similar for northern water snake *Nerodia sipedon* (fenced: 5; unfenced: 4). Ten fenced and ungrazed and 10 unfenced and grazed streams and riparian areas (100 m long, 10–15 m wide) were compared on private farms. All ungrazed sites had been grazed until they were fenced to exclude livestock 1–2 years prior to 1998 (4-strand electric fence). Unfenced stream sites and surrounding pastures were grazed continuously with an average stocking rate of 0.4 animals/ha. Reptiles were monitored using drift fences with pitfall traps set perpendicular to streams, coverboards and opportunistically using hand captures. Traps were checked 3–4 times a week from April–July 1998 and 1999.

A replicated, controlled, before-and-after study in 1994–1997 in three sites of chenopod scrubland in South Australia, Australia (7), found that overall reptile species richness and capture rates were similar at ungrazed sites compared to those under short-term intensive grazing, but that capture rates of one species increased one year after intensive grazing. Overall reptile captures and species richness were similar in an ungrazed area and in paddocks with short-term intensive grazing, both immediately before and after grazing, and one year after grazing (results reported as statistical model outputs). Central netted dragon *Ctenophorus nuchalis* capture rates remained similar in the ungrazed area (0.5–0.6 individuals/plot) and grazed areas immediately before and after grazing (before: 0.3–0.6 individuals/plot, after 0.5–0.5), but were lower one year later in the ungrazed area (0.4) than in the grazed area (1.1–1.5). See paper for details of other species capture rates. Reptiles were surveyed in three sites: an ungrazed area and two adjacent short-term intensively grazed paddocks (20 ha each). Intensive grazing consisted of releasing 70–80 cattle into each paddock for 6–18 days in winter and summer 1995. Reptile surveys took place twice before, twice immediately after, and twice one year after grazing using drift fences with pitfall traps open for 10 days at a time (18 fence-trap plots in grazed and in 12 in ungrazed paddocks). Captured lizards were marked with unique toe clips.

A site comparison study in 1999–2000 in grazed and cultivated semi-stable sand dunes in Zaranik Protected Area in North Sinai, Egypt (8) found that excluding livestock grazing increased Be'er Sheva fringe-fingered lizard *Acanthodactylus longipes* abundance. Fringe-fingered lizards were more than three times as abundant in ungrazed fenced (29 individuals/site) compared to unfenced grazed sites (9 individuals/site). Lizards spent less time moving and were observed further away from the nearest vegetation in ungrazed fenced compared to unfenced grazed sites (60 vs 38 seconds, 105 vs 55 cm). Lizards were sampled in three sites protected by fences in a protected area and three unfenced sites subject to grazing and low-impact watermelon farming. All sites were 50 m x 50 m. Sites were sampled two to four times monthly, between September 1999 and September 2000. Lizards were visually observed for three minutes and captured for measurement and marking when possible.

A paired sites, controlled study in 2002–2003 of semi-desert shrub and grassland, south-eastern Arizona, USA (9) found that lizard species richness was similar in ungrazed and grazed sites, but that some species abundances were higher in ungrazed sites, depending on the vegetation type. Species richness was the same in ungrazed and grazed sites (both 7–8 species). In tarbrush-dominated vegetation, two species were more abundant in ungrazed (eastern fence lizards *Sceloporus undulatus*: 17 individuals; common side-blotched lizards *Uta stansburiana*: 21) than grazed land (eastern fence: 2; side-blotched: 1), one species was less abundant (round-tailed horned lizards *Phrynosoma modestum* ungrazed: 3, grazed: 13) and one species had similar abundances (western whiptail *Cnemidophorus tigris*: 31, 37). For three species sample sizes were too small for analysis (desert spiny *Sceloporus magister*: 0, 1; ornate tree *Urosaurus ornatus*: 2, 1; grassland whiptail lizards *Cnemidophorus uniparens*: 7, 3). In creosote-dominated vegetation, four of eight lizard species abundances were similar in ungrazed and grazed land (eastern fence 26, 17; side-blotched 34, 29; round-tailed horned: 10, 11; western whiptail: 85, 82). For four species, sample sizes were too small for analysis (desert spiny lizard: 2, 3; ornate tree lizard: 3, 4; western banded gecko *Coleonyx variegatus*: 1, 1; grassland whiptail: 8, 2). A 9 ha area was fenced (post and barbed wire) to exclude livestock in 1958. Grazing continued outside of the enclosure. Lizards were monitored using pitfall traps along 12–13 transects (3–5 traps/transect) that extended from outside to inside the enclosure (60–100 m each side of the enclosure, 20–250 m apart) in August 2002 (728 trap nights) and May–August 2003 (4,620 trap nights). Transects included two vegetation types: tarbrush (1,428 trap nights) and creosote (3,920 trap nights). All lizards, other than western banded geckos, were individually marked with toe clips. Only adults were included in the analysis.

A paired sites, controlled, before-and-after study in 1993–1996 and 2007 in chenopod scrubland in South Australia, Australia (10) found that the effect of ceasing grazing on abundance varied depending on the species. After fencing to exclude livestock, one gecko species increased (knob-tailed gecko *Nephrurus levis* after fencing: 3.3 individuals/plot; before fencing: 0.3–0.5 individuals/plot) and two geckos decreased (tessellated gecko *Diplodactylus tessellatus* after fencing: 0.0; before fencing: 1.2–1.7; variable fat-tailed gecko *Diplodactylus conspicullatus* after fencing: 0.4; before fencing: 1.5–1.9) in abundance compared to beforehand when the same plots were grazed. The abundance of five other species remained similar after grazers were excluded (see paper for details). Four paired sites of differing grazing pressure were set out in 1993 (low intensity grazing: <12 cattle dung/ha; medium: 12–100; high: >120). After four years, three of the eight grazing pressure sites were fenced to exclude cattle and predators. Reptiles were sampled for 10 days in summer from 1993–1996 and 2007 using 300 mm long flymesh drift fences with 13 unbaited pitfall traps (500 mm deep x 150 mm wide, 8 m apart).

A replicated, controlled study in 1997–2006 in scrub and grassland in central California, USA (11) found the abundances of one of three reptiles increased more slowly in ungrazed plots compared to grazed plots following burning (natural and prescribed). The abundance of blunt-nosed leopard lizards *Gambelia sila* increased at a slower rate in ungrazed plots (1 extra individuals/year) compared to grazed plots (7 extra individuals/year). The change in abundances of two other species (western whiptail lizards *Aspidoscelis tigris* and side-blotched lizards *Uta stansburiana*), and the overall abundances of all three species did not differ between ungrazed and grazed plots (see original paper for details). Four 3 km^2 areas in a single site were established and grazed from December–April in 1998–2001 and 2005–2006. Grazing intensity varied between years and the whole site had been burned (natural and prescribed fire) in 1997. Within each area, a 25 ha plot was fenced to exclude livestock (ungrazed). Day-active lizards were surveyed visually within a 9 ha grid in each grazed and ungrazed area on 10 days in May–July in 1997–2006 (800 survey days).

A replicated, randomized, site comparison study in 2011 in desert shrub and grassland in the western Mojave Desert, California, USA (12) found that Agassiz's desert tortoises *Gopherus agassizii* were more abundant and had a lower mortality rate in a protected area fenced to exclude livestock grazing and recreational vehicle use. Desert tortoise densities were approximately six-times higher in the most protected area, the Tortoise Natural Area (15 tortoises/km^2) than in designated tortoise critical habitat (2 tortoises/km^2) and four-times higher than on private lands (4 tortoises/km^2). Tortoise annual death rates over the preceding four years were estimated as lowest in the Tortoise Natural Area (3%/year) compared to private lands (6%/year) or in critical habitat (20%/year, results not statistically compared). Tortoises were surveyed in 240 1 ha plots across three different management areas (80 plots/area): Tortoise Natural Area (1973: closed to recreational vehicles; 1980: fully enclosed and closed to mining and livestock grazing, 2010: 12 km of fencing extended to prevent tortoises leaving), tortoise critical habitat areas (1994: recreational vehicle use restricted but not enforced with some annual closures, 1990: closed to sheep grazing) and private lands (unregulated sheep grazing, intensive recreational vehicle use, hunting and rubbish dumping). In April–May 2011 plots were surveyed on foot twice in a day for live or dead tortoises and field signs.

A controlled, before-and-after study in 1997–2013 in an area of mixed dry and wet heathland in Dorset, UK (13, same experimental set-up as 14) found that an area where grazing cattle were excluded with a fence had more smooth snakes *Coronella austriaca* compared to an area where grazing continued. Over four years after grazing cattle were excluded, more smooth snakes were found in the ungrazed area (28 snakes) compared to the grazed area (16 snakes). During the previous 13 years when the whole area was grazed, the number of snakes caught in each area was similar during 12 of 13 years (3–8 snakes/year). In February 2010, a fence was erected to exclude cattle from a 6 ha area of heathland that had been grazed by cattle *Bos taurus* during May–September since 1997 (0.1–0.3 cows/ha). The remaining 4 ha continued to be grazed after the fence was erected. In 1997–2013, annual surveys for reptiles were conducted (21 surveys/year, though only 18 in 1997 and three in 2002) by randomly placing groups of 37 artificial refuges (corrugated steel sheets) in a hexagonal pattern (5–7 groups of refuges in the ungrazed area; four groups in the grazed area). All refuges were checked for reptiles, and smooth snakes were individually marked using PIT tags.

A controlled study in 2010–2013 in an area of mixed dry and wet heathland in Dorset, UK (14, same experimental set-up as 13) found that three of four reptile species were more abundant in ungrazed compared to grazed areas, and the fourth species occurred at similar numbers in both areas. The ungrazed area contained more grass snakes *Natrix natrix* (2/plot), slow worms *Anguis fragilis* (67/plot) and common lizards *Zootoca vivipara* (13/plot) than the grazed area (grass snakes: 1/plot; slow worms: 29/plot; common lizards: 6/plot), whereas a similar number of sand lizards *Lacerta agilis* were found in the ungrazed (3/plot) and grazed (6/plot) areas. In February 2010, a fence was erected to exclude cattle from a 6 ha area of heathland that had been grazed by cattle *Bos taurus*. The remaining 4 ha continued to be grazed after the fence was erected. In 2010–2013, annual surveys for reptiles were conducted (21 surveys/year) by randomly placing 11 groups of 37 artificial refuges (407 refuges in total) during April–October (seven groups of refuges in the ungrazed area; four groups in the grazed area). The number of reptiles of each species was recorded at each visit.

A replicated, site-comparison study in 1997–2007 in shrub and woodland in south eastern Australia, Australia (15) found that ungrazed and grazed sites had similar combined reptile and small mammal species richness. Over 11 years, reptile and small mammal species richness remained similar in ungrazed (0.03 species/100 trap nights/year) and grazed shrubland (0.04 species/100 trap nights/year). Over the same time period, livestock removal did not affect the change in overall reptile and small mammal abundance over time in shrubland (no livestock: 0.02 individuals/100 trap nights/year; with livestock: 0.11) In 1997–2007, reptiles and small mammals were surveyed in two shrubland sites (degraded chenopod shrubland dominated by *A. victoriae*.) with historical but no current domestic livestock grazing and two sites with livestock (sheep and/or cattle) grazing. Reptiles were surveyed using pitfall traps one–three times/year (22 surveys).

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Forest, open woodland & savanna

- **Five studies** evaluated the effects of ceasing livestock grazing in forest, open woodland and savanna on reptile populations. Two studies were in each of Argentina[2,4] and Australia[3,5] and one was in Mexico[1].

COMMUNITY RESPONSE (4 STUDIES)

- **Richness/diversity (4 studies):** Three of four studies (including two replicated, site-comparison studies) in Mexico[1], Argentina[2,4] and Australia[5] found that ungrazed and grazed areas, in one case with burning[4], had similar reptile species richness[1,4] and diversity[2,4]. The other study[5] found that in areas where livestock grazing was stopped, combined reptile and small mammal species richness increased more than in areas with grazing.

POPULATION RESPONSE (5 STUDIES)

- **Abundance (5 studies):** Two of five studies (including three replicated, site comparison studies) in Mexico[1], Argentina[2,4] and Australia[3,5] found that ungrazed areas had a higher abundance of reptiles[3] and lizards[1] than grazed areas. Two studies[4,5] found that ungrazed areas, in one case with burning[4], had similar overall reptile[4] or reptile and small mammal[5] abundance compared to grazed areas. The other study[2] found that grazing had mixed effects on reptile abundance.

BEHAVIOUR (0 STUDIES)

A paired, controlled study in 1991 of tropical deciduous forest ranchland in Baja California Sur, Mexico (1) found that lizard abundances tended to be higher in ungrazed sites compared to grazed sites. Results were not statistically tested. Thirty-two lizards were observed in ungrazed sites and seven in grazed sites. Five species were observed in both ungrazed and grazed sites: spiny lizard *Sceloporus hunsakeri* (ungrazed: 6 individuals, grazed: 2), Baja California brush lizard *Urosaurus nigricaudus* (16, 2), orange-throated whiptail *Aspidoscelis hyperythrus hyperythrus* (7, 1), spiny lizard *Sceloporus licki* (2, 1), and Baja blue rock lizard *Petrosaurus thalassinus thalassinus* (1, 1). Five 25 x 5 m transects at 5 m intervals were established in a 2,400 m² exclosure with no grazing since 1989. The same survey set up was established in a grazed area 35 m outside the exclosure on a livestock ranch. Lizard abundance was measured by counting the number of lizards observed/time spent looking.

A replicated, site comparison study in 1994–1998 in woodland savanna near Santo Domingo, Argentina (2) found that 25 years after cattle were excluded, overall snake and lizard abundances and diversity tended to be similar to adjacent grazed ranchland. A total of 82 snakes of 15 species and 136 lizards of 12 species were captured in ungrazed land compared to 71 snakes of 16 species and 182 lizards of 10 species in grazed land (results were not statistically tested). Species diversity was similar between ungrazed restored and grazed land (Shannon Wiener Diversity index of snakes ungrazed: 2.4, grazed: 2.4; lizards: 1.7, 1.6). One lizard and one snake species were more abundant in ungrazed land, and two lizards and one snake species were more abundant in grazed land (see paper for details). Reptiles were monitored in an area fenced in 1976 to exclude cattle and allow woodland regeneration (10,000 ha) and an adjacent overgrazed ranchland (7,500 ha). Surveys were carried out in six plots of each habitat type (>7 km apart) using drift fences with funnel traps ('arrays', 6 traps/array, one array/plot) in March 1994–March 1998 (152 non-consecutive days).

A replicated, site comparison study in 2001 in savanna woodland in Queensland, Australia (3) found that overall reptile abundance and the abundance of five of 18 species was higher in ungrazed than grazed plots. Overall reptile abundance was higher in ungrazed (18.5–19.6 individuals/plot) than grazed plots (12.3–14.0), regardless of fire history. Of 32 reptile species observed, 18 were included in analysis (appeared in high enough numbers). Five species abundances were higher in ungrazed than grazed plots (eastern bearded dragons *Pogona barbata* ungrazed: 0.6–0.7 individuals/plot vs grazed: 0–0.1; variable fat-tailed geckoes *Diplodactylus conspicillatus*: 0.8–1.0 vs. 0.1–0.2; stout ctenotus *Ctenotus hebetior*: 2.6–4.3 vs. 2.0–2.3; leopard ctenotus *Ctenotus pantherinus*: 1.4–4.4 vs. 0–1.3, red-earth ctenotus *Ctenotus rosarium*: 1.9–2.0 vs. 1.0–1.3). Dwarf skink *Menetia greyii* abundance was lower in ungrazed (0–0.3) than grazed plots (1.0–1.3). The abundance of the remaining 12 species was similar in ungrazed and grazed plots. In January 2001, reptiles were monitored on three cattle stations (>20,000 ha each) in 29 one-ha plots that were either ungrazed (paddocks where cattle were excluded) or grazed (4–8 cattle/ha). Plots were also either recently burned (within 2 years) or unburned (last burnt >2 years ago). Reptiles were sampled using cage traps and pitfalls supplemented by day and night log rolling and litter raking.

A site comparison study in 2006 of cattle pasture in Corrientes, Argentina (4) found that overall reptile diversity, species richness and abundance were similar in ungrazed sites (with annual fires or no fire for three or 11 years) and grazed sites with annual prescribed fires. Overall reptile species richness, abundance and diversity were similar in ungrazed sites that had either annual fires or no fires for three or 11 years (richness: 3–4; abundance: 22–44, Shannon diversity index: 0.8–1.1) compared to grazed sites with annual prescribed fires (richness: 4; abundance: 17, Shannon diversity index: 1.1). Species composition was most similar in sites that were ungrazed with annual fires and sites that were grazed with annual fires (result reported as similarity index). Four areas (≥ 400 ha) were monitored: ungrazed and no fires for three years; no grazing or fires for 11 years;

ungrazed with annual fires (August–September); grazed (3 ha/cattle unit) with annual fires. Monitoring was undertaken using drift-fencing with pitfall traps in January–April 2006 (80 survey days).

A replicated, site comparison study in 1997–2007 in open woodland in south eastern Australia, Australia (5) found that following removal of domestic livestock, combined reptile and small mammal species richness, but not abundance, increased. Over 11 years, overall reptile and small mammal species richness increased after livestock removal in woodland (0.04 species/100 trap nights/year) compared to areas with livestock (0.01 species/100 trap nights/year). Over the same time period, livestock removal did not affect the change in overall reptile and small mammal abundance over time (no livestock: −0.40 individuals/100 trap nights/year; with livestock: −0.31). In 1997–2007, reptiles and small mammals were surveyed in two woodland sites (open mulga *Acacia aneura* woodland) with historical but no current domestic livestock grazing and two sites with livestock (sheep and/or cattle) grazing in the Flinders Ranges. Reptiles were surveyed using pitfall traps one–three times/year (23 surveys).

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Wetland

- **Two studies** evaluated the effects of ceasing livestock grazing in wetlands on reptile populations. One study was in the USA[1] and one was in Australia[2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Abundance (2 studies):** One replicated, site comparison study in the USA[1] found that ungrazed sites had fewer bog turtles than grazed sites. One replicated, randomized, controlled study in Australia[2] found that ungrazed areas had similar overall reptile and amphibian abundance compared to that were grazed, burned or grazed and burned (to remove invasive non-native para grass). The study[2] also found that unmanaged areas (no grazing or burning) had a higher abundance of one skink species than areas with grazing and/or burning.

- **Occupancy/range (1 study):** One replicated, site comparison study in the USA[1] found that juvenile box turtles were present less frequently in ungrazed sites compared to grazed sites.

BEHAVIOUR (0 STUDIES)

A replicated, site comparison study in 2000–2001 in wet meadow or fen areas on farmlands in New Jersey, USA (1) found that ungrazed areas had fewer bog turtle *Glyptemys muhlenbergii* captures and densities and lower occurrence of juvenile turtles compared to grazed sites. Overall bog turtle captures and density in formerly grazed sites (captures: 3 individuals/site; density: 8 turtles/ha) was lower than in currently grazed sites (6, 25). Juvenile turtles were found less frequently in formerly sites (33%) compared to currently grazed sites (75%). Each hectare of 12 formerly grazed (no livestock for at least 10 years) and 12 grazed (under constant grazing for >50 years; 11 grazed by cattle, one by horses) sites were visually searched for a total of 15 hours over at least three visits in April–September 2000–2001. All captured turtles were sexed, measured, marked by notching shells and released at site of capture.

A replicated, randomized, controlled study in 2004–2006 in a seasonal wetland in Queensland, Australia (2) found that overall reptile and amphibian abundances were similar in ungrazed areas compared to areas that were grazed, burned or grazed and burned (to control invasive para grass *Urochloa mutica*), but that abundance of one skink species *Lampropholis delicata* was reduced in areas with grazing and/or burning. Overall reptile and amphibian abundance was similar in ungrazed areas compared to areas that were grazed, burned or grazed and burned (results presented as statistical model outputs). However, abundance of *Lampropholis delicata* was higher in ungrazed plots with no burning (14 skinks/plot) compared to plots with grazing and/or burning (grazed: 4 skinks/plot; burned: 3 skinks/plot; grazed and burned: 1 skink/plot). Para-grass dominated habitat in a conservation park (3,245 ha) was divided into 12 plots (200 x 300 m each) and each plot was either left unmanaged (no grazing or burning), grazed, burned, or grazed and burned (3 plots/management type). Burning took place in August 2004, September 2005 and November 2006. Cattle *Bos indicus* grazing took place after burning in September–December 2004, October–December 2005 and November–December 2006. Livestock levels were calculated to consume 50% of the grass biomass present/plot. Reptile and frog communities were sampled four times between 2005–2007 using three pitfall/funnel trap arrays/plot (see original paper for details). Reptiles were individually marked by toe clipping prior to release.

(1) Tesauro J. & Ehrenfeld D. (2007) The effects of livestock grazing on the bog turtle [*Glyptemys* (=*Clemmys*) *muhlenbergii*]. *Herpetologica*, 63, 293–300.
(2) Bower D.S., Valentine L.E., Grice A.C., Hodgson L. & Schwarzkopf L. (2014) A trade-off in conservation: Weed management decreases the abundance of common reptile and frog species while restoring an invaded floodplain. *Biological Conservation*, 179, 123–128.

3.7. Raise mowing height

- **One study** evaluated the effects of raising mowing height on reptile populations. This study was in Australia[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (1 STUDY)

- **Behaviour change (1 study):** One replicated, site comparison study in Australia[1] found that in long-sward pastures or crops marbled geckos did not navigate directly towards a tree, whereas in short-sward pastures they did.

Background
Vegetation height is important in determining the value of cropland and grassland to wildlife. High vegetation can provide more complex environments and more habitats, while short vegetation may increase the risk of predation. Cutting of crops or pasture can kill or maim reptiles causing mutilations, crushing, and other lethal injuries. It is possible that mortalities and injuries could be reduced by raising the height of cutting machinery above the soil (Saumure *et al.* 2007).

Saumure, R.A., Herman, T.B., & Titman, R.D. (2007) Effects of haying and agricultural practices on a declining species: The North American wood turtle *Glyptemys insculpta*. *Biological Conservation*, 135, 565–575.

A replicated, site comparison study in 2014 in mixed crop and pastureland in south-eastern Australia (1) found that marbled geckos *Christinus marmoratus* did not navigate directly towards trees in tall-sward pastures, but did in shorter sward pasture. In long native or exotic pastures or in wheat *Triticum vulgare* or canola *Brassica napus* crops, marbled geckos did not orient directly towards a target tree, but did in short native or exotic pasture (results reported as statistical model outputs, see original paper for details). Individual wild arboreal geckos were released into fields with an isolated tree surrounded by different pasture or crop fields and direction of travel was recorded. The field types included long (average sward height >20 cm) and short (average sward height <10 cm) pastures dominated by either native or exotic plants, or one of two cereal crops (wheat or canola; 6 total field types). Lizards were released in three fields/type (>2 km apart; 18 total fields). Geckos were caught from the same landscape but >5 km away from the study site. Individual animals were marked with fluorescent powder and tracked for 6 hours after release.

(1) Kay G.M., Driscoll D.A., Lindenmayer D.B., Pulsford S.A & Mortelliti A. (2016) Pasture height and crop direction influence reptile movement in an agricultural matrix. *Agriculture, Ecosystems and Environment*, 235, 164–171.

3.8. Create uncultivated margins around arable or pasture fields

- **Two studies** evaluated the effects of creating uncultivated margins around arable or pasture fields on reptile populations. One study was in Australia[1] and one was in the UK[2]

COMMUNITY RESPONSE (1 STUDY)

- **Richness/diversity (1 study):** One replicated, site comparison study in Australia[1] found that revegetated linear strips had similar reptile species richness compared to cleared and remnant strips. The study[1] also found that revegetated strips and patches had similar reptile species richness.

POPULATION RESPONSE (1 STUDY)

- **Abundance (1 study):** One replicated, site comparison study in Australia[1] found that revegetated linear strips had similar reptile abundance compared to cleared and remnant strips. The study also found that revegetated strips and patches had similar reptile abundance.

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One replicated study in the UK[2] found that uncultivated field margins were used by slow worms, common lizards and grass snakes, but not by adders.

Background

Creating margins around agricultural fields may provide important habitat for a range of reptile species. This intervention includes both cases where field margin vegetation is allowed to regenerate naturally, as well as cases where margins are planted/sown with desirable plant species.

This action does not cover hedgerows, which are included in *Provide or maintain hedgerows on farmland*. See also *Provide or retain set-aside areas on farmland*.

For studies on the effect of different actions for managing existing vegetation, see *Raise mowing height*; *Habitat restoration and creation – Manage vegetation using herbicides; Manage vegetation by cutting or mowing*; *Manage vegetation by hand* and *Manage vegetation using livestock grazing.*

A replicated, site comparison study in 2008–2009 on agricultural land in Victoria, Australia (1) found that revegetating linear habitat strips did not increase reptile species richness and abundance compared to cleared or remnant strips of habitat, nor was there a difference between revegetating in strips or patches. Revegetated linear strips had similar reptile richness and abundance (richness: 0.1–0.5 species/strip, abundance: 0.1–0.4 individuals/strip) compared to cleared (0.2–0.3, 0.3–0.4) and remnant strips (0.4–0.5, 0.3–0.5). Revegetated linear strips also had similar richness and abundances to revegetated patches (data not reported). Reptiles were monitored in five locations in each of two regions on or bordering agricultural land. Drift fences with pitfall traps were set out in sites classified as: revegetated linear strip (using native plants 8–14 years before), cleared linear strip, remnant linear strip, and remnant patches (10 traps/site). Surveys were carried out for five consecutive days/month in January–March 2008 and 2009. Remnant patches and enlarged remnant patches revegetated with native vegetation were also surveyed in five different locations in the same two regions using the same methods.

A replicated study in 2014–2015 on 14 farms in the UK (2) found that uncultivated field margins were used by slow worms *Anguis fragilis*, common lizards *Zootoca vivipara* and grass snakes *Natrix Helvetica* but not by adders *Vipera berus*. From two separate surveys, uncultivated margins were occupied by slow worms (occupied 8% and 14% of surveyed margins), common lizards (occupied 5% and 32% of surveyed margins) and grass snakes (occupied 45% and 49% of surveyed margins), but adders were not found in any margins. One analysis method showed that slow worms and grass snakes were found less frequently in margins with taller vegetation, and common lizards were found more frequently in wider margins with deeper ditches (presented as model result; see paper for more details). In 2014, ten farms were selected and eight 100 m transects were established in uncultivated field margins on each farm (south, east or facing margins only). In 2015, a total of 80 transects (100m) were established on margins across six farms (facing any direction). Five groups of 2–3 refuges (roofing felt/corrugated sheets and carpet tiles; 50 x 50 cm) were set at 20 m intervals along the transects. Transects were searched 12–15 times in 2014 and 6–10 times in 2015 during April–November. Presence or absence of reptiles was recorded.

(1) Jellinek S., Parris K.M., McCarthy M.A., Wintle B.A. & Driscoll D.A. (2014) Reptiles in restored agricultural landscapes: the value of linear strips, patches and habitat condition. *Animal conservation*, 17, 544–554.

(2) Salazar R., Foster J. & Thompson P. (2016) *Evaluating the importance of agri-environment scheme buffer strips to widespread amphibians and reptiles* [Environmental Stewardship Monitoring and Evaluation Framework Reference ECM6147]: Final report. Report to Natural England.

3.9. Provide or maintain hedgerows on farmland

- **One study** evaluated the effects of providing or maintaining linear features on reptile populations. This study was in Madagascar[1].

COMMUNITY RESPONSE (1 STUDY)

- **Community composition (1 study):** One site comparison study in Madagascar[1] found that reptile communities in cultivated areas with hedges were more similar to those found in forests than were communities from cultivated areas without hedges. The study[1] also found that more reptile species were found only areas with hedges than only in areas without hedges.

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (0 STUDIES)

Background

Agricultural intensification, including increases in field sizes and pesticides use, has resulted in a loss of field boundary habitats, such as hedgerows. These features can provide a relatively undisturbed habitat for wildlife in intensively managed agricultural landscapes. Hedge planting and maintenance of existing hedges has, therefore, been proposed as a means of preserving and enhancing biodiversity. Such management is sometimes funded through agri-environmental schemes.

This action does not include studies on the effect of uncultivated margins, which are included in *Create uncultivated margins around arable or pasture fields*.

A site comparison study in 2012 in two sites of tropical dry forest and farmland in south-western Madagascar (1) found that a site with hedges throughout different habitats had smaller differences in reptile communities than those without hedges, and that cultivated areas with hedges had more species than cultivated areas without hedges. The similarity of reptile communities in cultivated areas, undegraded forest and degraded forest was higher in the site with hedges than in the site without hedges (result reported as a dissimilarity index). Nine species were found in cultivated areas with hedges (1–19 individuals) that were not found in cultivated areas with no hedges, whereas the opposite was true for only two species (1–3 individuals). Two sites were selected that contained undegraded forest, degraded forest and cultivated areas. In one site, hedges (2 m high, containing non-native *Opuntia* spp. and native vegetation e.g. *Euphorbia stenoclada*) surrounded cultivated areas and bordered degraded forest. The other site had no hedges. Eight 100 m transects were established in each habitat, and all reptile species were recorded within 1.5 m of the transect line (10 surveys in February–April 2012). In cultivated areas transects followed field boundaries with or without hedging.

(1) Nopper J., Lauströer B., Rödel M.O. & Ganzhorn J.U. (2017) A structurally enriched agricultural landscape maintains high reptile diversity in sub-arid south-western Madagascar. *Journal of Applied Ecology*, 54, 480–488.

3.10. Provide or retain set-aside areas on farmland

- We found no studies that evaluated the effects of providing or retaining set-aside areas on farmland on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background
Allocation of some farmland to set-aside (fields taken out of production) was compulsory under European Union agricultural policy from 1992 until 2008. The idea was to reduce production. However, set-aside has also been promoted as a method of enhancing biodiversity on farmland. Set-aside can be rotational (in a different place every year or two) or non-rotational (same place for 5–20 years) and fields can either be sown with fallow crops or left to naturally regenerate. Unlike fallow land, set-aside is not ploughed or harrowed except for the purpose of sowing. However, set-aside often is managed by cutting and/or spraying. In some cases, set-aside land has had wildflowers sown on it.

For studies on the effect of different actions for managing existing vegetation, see *Raise mowing height*; *Habitat restoration and creation – Manage vegetation using herbicides; Manage vegetation by cutting or mowing*; *Manage vegetation by hand* and *Manage vegetation using livestock grazing.*

3.11. Prevent access to livestock water feeders

- **One study** evaluated the effects of preventing access to livestock water feeders on reptile populations. This study was in Morocco[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Survival (1 study):** One replicated, randomized, controlled, before-and-after study in Morocco[1] found that covering water feeder openings with wire mesh resulted in fewer combined reptiles and amphibians being trapped compared to water feeders without mesh covers.

BEHAVIOUR (0 STUDIES)

Background
Reptiles can become trapped in low level water feeders or feeders with vertical sides. Preventing access to feeders by using mesh covers may reduce reptile mortality.

A replicated, randomized, controlled, before-and-after study in 2014–2016 in three areas of desert shrub and grassland in south-western Morocco (1) found that adding mesh covers to ground-level livestock water feeders resulted in fewer reptiles and amphibians being trapped. Ground-level water feeders with in and outflow openings covered by wire mesh trapped fewer reptiles and amphibians (17% of feeders; 0.3 individuals trapped/feeder) compared to water feeders without wire mesh covers attached (36% of feeders; 0.8 individuals trapped/feeder). Before mesh covers were attached, numbers of reptiles and amphibian species trapped were similar amongst all feeders located in the same areas (see original paper for details). Rectangular concrete, roofed, ground-level water feeders ('cisterns', all <15 years old) designed to capture rainwater for livestock were surveyed in three zones (168–212 km^2) in the north-western Sahara Desert (24 feeders/zone). Water inlet and overflow openings were covered with wire mesh on 12 randomly selected feeders/zone. All feeders were surveyed for reptiles and amphibians once before mesh was applied in November 2014 (zones 1 and 2) or October 2015 (zone 3) and again either four times in June 2015–April 2016 (zones 1 and 2) or twice in March–April 2016 (zone 3). All dead and live reptiles and amphibians found inside feeders were recorded.

(1) Pleguezuelos J.M., García-Cardenete L., Caro J., Feriche M., Pérez-García M.T., Santos X., Sicilia M. & Fahd S. (2017) Barriers for conservation: mitigating the impact on amphibians and reptiles by water cisterns in arid environments, *Amphibia-Reptilia*, 38, 113–118.

3.12. Retain or increase leaf litter or other types of mulch

- **Two studies** evaluated the effects of retaining or increasing leaf litter or other types of mulch on reptile populations. One study was in Indonesia[1] and one was in Australia[2].

COMMUNITY RESPONSE (1 STUDY)

- **Richness/diversity (1 study):** One replicated, randomized, controlled, before-and-after study in Indonesia[1] found that reptile species richness increased with the addition of leaf litter and decreased following removal of leaf litter and woody debris.

POPULATION RESPONSE (2 STUDIES)

- **Abundance (2 studies):** Two randomized, controlled studies (one replicated, before-and-after study) in Indonesia[1] and Australia[2] found that the addition of leaf litter or cacao husks resulted in a higher abundance of overall reptiles[1] or skinks[2]. One study[1] also found that removal of leaf litter and woody debris led to a decrease in reptile abundance.

BEHAVIOUR (0 STUDIES)

Background
The presence of leaf litter is an important driver of reptile abundance in both temperate (Hu *et al.* 2013) and tropical forests (Heinen 1992). Adding leaf litter

or other mulch to woody crops such as cacao may provide habitat for reptile communities.

For studies that leave woody debris after logging activities or add woody debris to the landscape see *Threat: Biological resource use – Leave woody debris in forests after logging*, and *Habitat restoration and creation – Add woody debris to landscapes*.

Heinen J.T. (1992) Comparisons of the leaf litter herpetofauna in abandoned cacao plantations and primary rain forest in Costa Rica: some implications for faunal restoration. *Biotropica*, 24, 431–439.

Hu Y., Urlus J., Gillespie G., Letnic M. & Jessop T.S. (2013) Evaluating the role of fire disturbance in structuring small reptile communities in temperate forests. *Biodiversity and Conservation*, 22, 1949–1963.

A replicated, randomized, controlled, before-and-after study in 2007–2008 in cacao plantations Sulawesi, Indonesia (1) found that both reptile abundance and species richness increased after the addition of leaf litter and decreased following the combined removal of woody debris and leaf litter. All results were reported as statistical model outputs. Overall reptile abundance increased when leaf litter was added but decreased after leaf litter and woody debris were removed, or when only woody debris was removed (see original paper for details of individual species abundance changes). Reptile species richness increased after leaf litter was added and decreased after leaf litter and woody debris were removed. Forty-two plots (40 x 40 m^2) in cacao plantations (number not specified) were randomly divided into seven treatments: removal and addition of leaf litter, removal and addition of woody debris (trunks and branch piles), removal and addition of woody debris plus leaf litter and no management (6 replicates of each treatment). Plots were sampled 26 days before and 26 days after habitat manipulation, three times a day in December 2007–July 2008. Active visual surveys were undertaken for 25 minutes along both plot diagonals (transects 3 x 113 m).

A randomized controlled study in 2014–2015 in a monoculture cacao farm in North Queensland, Australia (2) found that adding cacao fruit husks underneath cacao trees increased population densities of skinks. Plots of cacao trees with cacao fruit husks had greater densities of skinks (1.1–3.3 skinks/plot) compared to plots without fruit husks (0.3–0.8 skinks/plot). The increased densities of skinks did not reduce the amount of fruit on trees (see original paper for details). The effect of adding cacao fruit husks to the base of trees was monitored on a 1.8 ha monoculture cacao farm. Fourteen plots (15 m apart) were randomly selected, each comprising two adjacent rows of four consecutive flower-bearing trees. In November 2014, seven of the plots had 280 kg of fresh cacao fruit husks left over from processing added underneath all trees (35 kg/tree). A further 15kg/tree of husks were added in December 2014 and January 2015. Visual surveys for skinks were conducted in the mornings every two weeks from December 2014–March 2015.

(1) Wanger T.C., Saro A., Iskandar D.T., Brook B.W., Sodhi N.S., Clough Y. & Tscharntke T. (2009) Conservation value of cacao agroforestry for amphibians and reptiles in South-East Asia: combining correlative models with follow-up field experiments. *Journal of Applied Ecology*, 46, 823–832.

(2) Forbes S.J. & Northfield T.D. (2017) Increased pollinator habitat enhances cacao fruit set and predator conservation. *Ecological Applications*, 27, 887–899.

3.13. Diversify ground vegetation and canopy structure in the habitat around woody crops

- **Two studies** evaluated the effects of diversifying ground vegetation and canopy structure in the habitat around woody crops on reptile populations. One study was in Puerto Rico[1] and the other was in Spain[2].

COMMUNITY RESPONSE (1 STUDY)

- **Richness/diversity (1 study):** One replicated, paired, site comparison study in Spain[2] found that olive groves with natural ground cover had higher reptile species richness and diversity than those with bare ground, but groves planted with a single species as ground cover had similar richness and diversity as those with bare ground.

POPULATION RESPONSE (2 STUDIES)

- **Abundance (2 studies):** One replicated, site comparison study in Puerto Rico[1] found that two of three lizard species were less abundant in shade-grown coffee plantations than in sun-grown plantations. One replicated, paired, site comparison study in Spain[2] found that olive groves with ground cover had more reptiles than groves with bare ground.

BEHAVIOUR (0 STUDIES)

Background

Agricultural intensification methods of large area woody crops (e.g. olives, almonds, coffee) has led to an increase in bare ground (through increased tilling or herbicide) and lack of overstorey surrounding the crop trees. This loss of soil, herbaceous covering and canopy layers has been implicated in losses of wildlife diversity in agricultural systems.

For example, coffee is a small tree or shrub that naturally grows in the forest understory and may be grown commercially under trees that provide shade. In the 1970s, sun-tolerant coffee plants were developed in response to fungal diseases and many plantations removed all canopy trees. These high-yield cultivation practices are considered unsustainable and many coffee plantations, are switching to shade-grown varieties.

Diversifying the habitats where woody crops are grown with respect to both ground and canopy vegetation may increase available habitat for reptile communities within these agricultural systems.

A replicated, site comparison study in 2000 in six coffee plantations in north-central Puerto Rico (1) found that two of three lizard species were less abundant in shade-grown than sun-grown coffee plantations. Puerto Rican crested anole *Anolis cristatellus* and barred anole *Anolis stratulus* were less abundant in shade-grown (crested: 1,642 individuals/m^2; barred: 294) than sun-grown coffee plantations (2,034; 631). Upland grass anoles *Anolis krugi* abundance was similar in shade-grown and sun-grown coffee plantations (shade: 411 individuals/m^2; sun: 384). Four further species were observed, but in too low numbers to assess population differences between plantation types. Yellow-chinned anole *Anolis. gundlachi* and emerald anole *Anolis evermanni* were mostly observed in shade-grown (yellow: 525 individuals observed; emerald: 241) rather than sun-grown coffee plantations (2; 6), whereas common grass anole *Anolis puchellus* tended to be less frequently observed in shade-grown compared to sun-grown coffee plantations (shade: 2 individuals observed; sun: 28). Puerto Rican giant anole *A. cuvieri* observations were the same in shade-grown and sun-grown coffee plantations (5 individuals observed in both). Lizard abundance was estimated using mark-resightings in 4–6 circular 400m^2 plots in three sun-grown (closely-spaced 2–3 m high sun tolerant coffee trees with dense foliage) and three shade-grown coffee plantations (irregularly-spaced 2–4 m high coffee (or banana or citrus) trees under a canopy of medium and tall shade trees) in March–May 2000. Each plot was sampled for four consecutive days in spring. Lizards were marked at a distance using tree-marking spray paint guns and latex house paint.

A replicated, paired, site comparison study in 2014–2015 in seven olive groves in Andalusia, Spain (2) found that olive groves with natural ground crop cover had greater reptile species richness, diversity and higher reptile counts than groves with bare ground, whereas planted crop ground cover had more reptile observations, but not richness or diversity than bare ground. Reptile observations were higher in olive groves with ground cover (natural cover: 10 individuals/site; planted cover: 8) than groves with bare ground (5). Species richness was highest in olive groves with natural ground cover (2 species/site) compared to planted ground cover (1) or bare ground (1). Species diversity was higher in natural cover crop (Shannon Index: 2) than bare ground (1), but similar to planted cover crops (1). Species diversity in planted cover crops was similar to bare ground. Reptiles were monitored in paired sites in seven olive groves: one site with ground cover (either natural herbaceous cover: 3 sites or planted single-species ground crops: 4 sites), and the other with bare ground (7 sites). Study plots were located within an olive-dominated landscape with almost no natural vegetation, either irrigated or unirrigated, and 10–100 years old. In May and July 2014–2015 reptiles were surveyed using two 1–2 km line transect censuses/site repeated on three warm sunny days. Each transect was surveyed for 30 minutes (336 total transects; 168/year).

(1) Borkhataria R.R., Collazo J.A. & Groom M.J. (2012) Species abundance and potential biological control services in shade vs. sun coffee in Puerto Rico. *Agriculture, Ecosystems and Environment*, 151, 1–5.

(2) Carpio A.J., Castro J., Mingo V. & Tortosa F.S. (2017) Herbaceous cover enhances the squamate reptile community in woody crops. *Journal for Nature Conservation*, 37, 31–38.

3.14. Plant trees on farmland

- **Two studies** evaluated the effects of planting trees on farmland to benefit reptiles. Both studies were in Australia[1,2].

COMMUNITY RESPONSE (2 STUDIES)

- **Richness/diversity (2 studies):** One replicated, paired sites study in Australia[2] found that pastures with tree plantings had similar rare reptile species richness compared to pastures with no trees, but that more rare species were present with 50% canopy cover compared to 5% cover. One replicated, site comparison study in Australia[1] found that farms with restoration planting (of native ground cover and trees) had lower reptile species richness than farms with remnant vegetation (of old growth woodland or natural regrowth).

POPULATION RESPONSE (1 STUDY)

- **Abundance (1 study):** One replicated, paired sites study in Australia[2] found that pastures with tree plantings had higher abundance of rare reptiles than pastures with no trees, and that rare reptiles were more abundant with 50% canopy cover compared to 5% cover.

BEHAVIOUR (0 STUDIES)

Background

Agricultural intensification, which includes increasing field size and pesticide use, has resulted in a loss of shelter and food resources for wildlife, such as that provided by areas of trees. These features can provide a relatively undisturbed habitat for wildlife in intensively managed agricultural landscapes. Tree planting may therefore diversify habitat availability and, in younger plantations, may also provide areas of longer uncut grass than is available elsewhere in the landscape.

A replicated, site comparison study in 2002–2005 on 46 farms in New South Wales, Australia (1) found that reptile species richness was lower on farms with 7–20-year-old restoration plantings compared to farms without restoration plantings. Reptile species richness was lower in restoration planting plots (1.5 species/plot) compared to remnant natural vegetation plots (2.0 species/plot), and lower on farms with restoration plantings (3.6 species/farm) compared to farms without restoration plantings (4.7 species/farm). Of 22 reptile species detected, 11 were not recorded in restoration plantings (see paper for individual species abundances). Twenty-three landscapes (10,000 ha circles) were defined and two farms/landscape were selected. Twenty-three farms contained restoration plantings and 23 did not. Restoration plantings were 7–20-years-old (native ground cover and trees), and were compared to areas of remnant natural vegetation (old growth woodland, self-seeded regrowth woodland or coppice regrowth woodland recovering from logging or fire). In spring 2002–2005, four 1 ha plots/farm (184 total plots, number taken from text) were surveyed for reptiles along transects using active searches (20 minutes x 1 ha) and point searches under artificial substrates (corrugated iron sheets, piles of offcut wood or sets of four roof tiles, two points/transect). On farms with restoration planting, three plots/site were in restored vegetation and one plot/site was in remnant vegetation.

A replicated, paired sites study in 2014–2015 in six grazing pastures in New South Wales, Australia (2) found that planting trees in pasture paddocks increased rare reptile species abundance but not rare species richness. Rare reptile abundance in tree-planted pasture was greater (0.9 individuals/paddock) than in pasture without trees (0.7 individuals/paddock). Rare species richness was statistically similar in tree-planted pasture (2.8 species/paddock) and pasture without trees (1.9 species/paddock). Rare species richness and abundance were associated with amounts of surrounding woody vegetation, such that the authors estimated there to be 2.6 more rare species and 5.7 more counts of rare reptiles in sites with 50% woody cover compared to sites with 5% woody cover within three km (see original paper for individual species responses). In January 2014–March

2015, reptiles were surveyed in six farms grazed by sheep *Ovis aries* or cattle *Bos Taurus* in paddocks directly adjacent to remnants of native open grassy woodland. On each farm, two transects (each 80 m long) were surveyed: grazed pasture and grazed pasture with linear tree plantings (10–25 m between linear plantings, *Eucalyptus* and *Acacia* species planted in the previous 30 years). Surveys were carried out using drift fences, pitfall traps and funnel traps set at 20, 50 and 80 m intervals. Surveys took place for 5 days at a time in austral spring–summer. Rare species were defined as those captured in ≤4 sites with <70 total captures.

(1) Cunningham R.B., Lindenmayer D.B., Crane M., Michael D. & MacGreggor C. (2007) Reptiles and arboreal marsupial response to replanted vegetation in agricultural landscapes. *Ecological Applications* 17, 609–619.

(2) Pulsford S.A., Driscoll D.A., Barton P.S. & Lindenmayer D.B. (2017) Remnant vegetation, plantings and fences are beneficial for reptiles in agricultural landscapes. *Journal of Applied Ecology*, 54, 1710–1719.

Aquatic habitat management

3.15. Manage ditches on farmland

- We found no studies that evaluated the effects of managing ditches on farmland on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Intensification of agricultural and other land management can result in loss of ditch biodiversity through activities such as mowing, grazing, and use of fertilizer and pesticides leading to water pollution. These may affect reptile populations. Ditch management practices can vary in terms of the frequency, season and technique used to clean or dredge ditches or may involve the maintenance of erosion control structures.

Marine and freshwater aquaculture

3.16. Install and maintain anti-predator systems around aquaculture that prevent entanglement of reptiles

- We found no studies that evaluated the effects on reptile populations of installing and maintaining anti-predator systems around aquaculture that prevent entanglement of reptiles.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background
Marine aquaculture (mariculture) includes the breeding, rearing, and harvesting of ocean-based aquatic plants and animals in ocean-based cages or spat lines. Marine aquaculture produces many of the shellfish (e.g. oysters, clams, mussels), prawn and shrimp, as well as salmon (Salmonidae spp.) and other marine fish consumed by humans. Leatherback turtles *Dermochelys coriacea* are known to have become entangled in mussel farm lines and spat lines in Canada resulting in fatalities and with the anchoring line of a mussel farm in the North Atlantic (Price *et al.* 2016). Lines made of stiff material have been proposed to prevent entanglement (Price & Morris 2013). Interventions to reduce bycatch in marine fishing nets are discussed in *Threat: Biological resource use*.

Freshwater aquaculture involves the breeding, rearing and harvesting of freshwater species in ponds. Freshwater aquaculture is used to produce commercial quantities of freshwater crayfish, prawns, turtles and fish (e.g. carp and trout). As with marine aquaculture, there is little documented evidence of how reptiles interact with these freshwater fish farms, although freshwater turtles undoubtedly compete with fish for food in these systems. Interventions to reduce bycatch in freshwater fishing nets are discussed in *Threat: Biological resource use*.

Price C.S. & Morris J.A. (2013) *Marine cage culture and the environment: twenty-first century science informing a sustainable industry*. NOAA Technical Memorandum NOS NCCOS 164.

Price C.S., Keane E., Morin D., Vaccaro C., Bean D. & Morris Jr J.A. (2016) *Protected species and longline mussel aquaculture interactions*. NOAA Technical Memorandum NOS NCCOS 211.

4. Threat: Energy Production and mining

Energy production (renewable and non-renewable) and mining can have substantial impacts on reptile populations through the destruction, pollution and use of habitats in preparation for and during operations. Most interventions involve restoration of previously mined land, which may be hampered by contamination of the ground water or soil resulting from mining operations. Several other interventions consider actions to reduce the impact of ongoing operations on reptiles or to reduce human-wildlife conflict in order that motivations to carry out lethal control of these species will also be reduced. Strategies for reptiles affected by energy production and mining often involve translocation of the animals from the footprint of the energy production or mining development site; this intervention is discussed in *Species Management*. For more general actions that relate to habitat restoration or addressing impacts of pollution, see chapters *Habitat restoration and creation* and *Threat: Pollution*.

4.1. Limit heavy vehicle use

- We found no studies that evaluated the effects of limiting heavy vehicle use on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Heavy vehicle traffic associated with mining is known to have direct impacts on wildlife, particularly through roadkill and the destruction (crushing or collapse) of burrows (Lovich & Ennen 2011). Indirect effects include soil compaction, which may limit the ability of reptiles to burrow. For studies on the effectiveness of actions that address the impact of road vehicles, see *Threat: Transportation service corridors*.

Lovich J.E. & Ennen J.R. (2011) Wildlife conservation and solar energy development in the desert southwest, United States. *BioScience*, 61, 982–992.

4.2. Leave/maintain/restore strips of undisturbed habitat between solar arrays

- We found no studies that evaluated the effects of leaving/maintaining/restoring strips of undisturbed habitat between solar arrays on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Solar arrays are large modular installations constructed with many subunits attached to concrete bases and placed in areas with little traffic and large amounts of sunlight, such as deserts. Reptile populations are disturbed, not only by the installation itself, but by habitat destruction. Leavings strips of undisturbed habitat between solar arrays may act as corridors and allow for reptile populations to successfully persist on large energy farms (Lovich & Ennen 2011).

Lovich J.E. & Ennen J.R. (2011) Wildlife conservation and solar energy development in the desert southwest, United States. *BioScience*, 61, 982–992.

4.3. Regulate temperature of water discharged from power plants

- **One study** evaluated the effects of regulating temperature of water discharged from power plants. This study was in the USA[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One study in the USA[1] reported that power plant water cooling canals were occupied by a population of American crocodiles.

Background

Warm water discharged from power plants into the ocean can change ocean temperatures and as a result affect local habitat use by marine reptiles, such as sea turtles, which may seek out areas of specific temperatures (Madrak *et al*. 2016). Regulating the temperature of water discharged would also protect against any negative impacts of power plant closure, which can cause rapid changes in local ocean temperatures.

Madrak S.V., Lewison R.L., Seminoff J.A. & Eguchi T. (2016) Characterizing response of East Pacific green turtles to changing temperatures: using acoustic telemetry in a highly urbanized environment. *Animal Biotelemetry,* 4, 22.

A study in 1983–1993 in a system of cooling canals for a power plant in Florida, USA (1) reported that the canals were occupied by a population of American crocodiles *Crocodylus acutus* and that the population grew over the study period. A total of 55 nests were found in the canals, and the number of nest/year increased in the period from 1986–1993 (1983: 3 nests; 1993: 11 nests). The number of non-hatchlings crocodiles increased by an average of 9% each year from 1983–1993, and authors estimated that the populations consisted of 24–30 non-hatchling crocodiles. The Turkey Point power plant site (2,388 ha) had a large canal system acting as a closed loop system for cooling water discharged from the plant. Water temperatures in the canals averaged 38°C, and salinity was 36 ppt. Eight other non-connected canals are also located adjacent to the cooling canals. In 1983–1993, night surveys were conducted of the whole area twice/week to monitor crocodile distributions and survival of hatchlings and juveniles. In 1984–1993, nest surveys, night surveys for hatchlings and periodical day surveys were conducted within the cooling canal system and hatchlings were individually marked.

(1) Brandt L.A., Mazzotti F.J., Wilcox, J.R., Barker Jr P.D., Hasty Jr G.L. & Wasilcwski J. (1995) Status of the American crocodile (*Crocodylus acutus*) at a power plant site in Florida, USA. *Herpetological Natural History*, 3, 29–36.

4.4. Restore former mining or energy production sites

- **Thirteen studies** evaluated the effects of restoring former mining or energy production sites on reptile populations. Nine studies were in Australia[1,3,4,6-9,12,13], two were in the USA[5,11], one was in Spain[2] and one was on Reunion Island[10].

COMMUNITY RESPONSE (8 STUDIES)

- **Community composition (4 studies):** Two of four site comparison studies (including two replicated studies) in Austalia[1,8,12] and Spain[2] found that restored mining areas hosted different reptile communities than unmined areas[8,12]. One study[2] found that reptile communities in the oldest restored areas were most similar to unmined areas. The other study[1] found that restored mining areas that were seeded or received topsoil had similar community composition compared to surrounding unmined forests.
- **Richness/diversity (5 studies):** Two replicated, site comparison studies and one review in Australia[3,9,13] found that restored mining sites had lower reptile species richness than unmined sites. One replicated, before-and-after, site comparison study in Spain[2] found that after restoration, reptile species richness increased steadily over a six-year period. One replicated, site comparison study in Australia[4] found that restored areas supported most of typical reptile species found in the wider habitat.

POPULATION RESPONSE (8 STUDIES)

- **Abundance (7 studies):** Five of six replicated, site comparison studies and one review in Australia[3,4,8,9,12,13] found that in restored mining areas reptiles tended to be less abundant than in unmined areas. The other study[13] found mixed effects of restoration on reptile abundance. One replicated, controlled study in Australia[6] found that restored areas that were thinned and burned 10–18 years after restoration began had higher reptile abundance than restored areas that were not thinned and burned.
- **Reproductive success (2 studies):** One review in Australia[9] found that one study reported reptiles breeding in restored mining areas. One study on Reunion Island[10] found that four of 34 and eight of 40 artificial egg laying sites in restored mining areas were used by Reunion day geckos nine months and two years after installation respectively.
- **Condition (1 study):** One review of restoration of mining sites in Australia[9] found that three of three studies indicated that reptile size or condition was similar in restored mines and undisturbed areas.

BEHAVIOUR (5 STUDIES)

- **Use (4 studies):** Three studies (including one replicated, site comparison study) in Australia[4] and the USA[5,11] found that restored mining areas were occupied by up to 14 snake, five turtle and one lizard species[5,11], or that generalist reptile species colonized restoration sites more quickly than did specialist species[4]. One replicated, controlled study in Australia[7] found that Napoleon's skinks reintroduced to a restored mining site all moved to an unmined forest within one week of release.
- **Behaviour change (1 studies):** One review of restoration of mining sites in Australia[9] reported that one of one studies indicated that there were changes in behaviour of lizards between restored mines and undisturbed areas.

Background

Restoration of former mining sites usually involves establishing native or non-native plants, often with the main aim of reducing erosion or reducing the concentration of pollutants (Wong 2003). Restoration may also benefit reptile species found in and around former mining sites by creating habitat conditions similar to those found prior to mining operations.

Studies relating to revegetation for habitat creation, including the use of coarse woody debris, are summarized under *Habitat restoration and creation*.

Wong M.H. (2003) Ecological restoration of mine degraded soils, with emphasis on metal contaminated soils. *Chemosphere*, 50, 775–780.

A site comparison study in 1978–1984 of restored sites within bauxite-mined jarrah forest in Western Australia (1, same experimental set-up as 4) found that most species recorded in unmined forest were also recorded in restored ex-mined forest. Results were not statistically tested. In total, 17 reptile species were recorded in replanted sites compared to 23 in the surrounding unmined forest. Restored sites that received fresh top soil, or that were heavily sown with native seed were more similar to healthy forest (result reported as a similarity index) and had a higher abundance of reptiles (132 and 136 individuals) than restored sites that received no top soil or seed (40 individuals). Three restoration sites were planted native eucalypt species. One of the sites was also sown with native seed, and another received fresh topsoil (see paper for more details on restoration). In 1978–1984, reptiles were monitored monthly in a wide range of restored areas and in surrounding unmined forest (number of survey locations not provided). More detailed studies were conducted between December 1980 and February 1981 in three restored areas (4.5–10 years since restoration activities) and four unmined forests (two healthy and two poor quality sites). Surveys involved pitfall trapping, live-traps and hand-collecting.

A before-and-after, site comparison study in 1988–1994 of spoil benches of a lignite mine in northwest Spain (2) found that reseeded and fertilized spoil benches that had been created were colonized by six reptile species. Species richness increased steadily with time since seeding. Species composition was most similar to that in unmined undisturbed plots in the oldest restored plots (10-years-old). Bocage's wall lizard *Podarcis bocagei* colonized unvegetated restored plots in the first year and grass snakes *Natrix natrix* appeared in drainage ditches from the third year. Other reptile species colonized from the fourth year onwards once the scrub layer was well developed. Spoil benches (60 ha) were created, planted with a slurry of pasture mix seeds and mulch and were fertilized in 1984–1994. Subsequent management was minimal. Monitoring was undertaken annually on a single 2 ha plot over the six years following seeding and in 1994 on 10 randomly selected 2 ha plots seeded 0–10 years previously. Three randomly selected, undisturbed plots close to the mine were also monitored in 1994. Surveys involved a total of 30 hours of visual searches in February–November.

A replicated, site comparison study in 2000–2002 of woodland and scrub at five mines in Western Australia, Australia (3) found that reptiles recolonized restored sites over three–nine years, although species richness and abundances were lower than on adjacent, undisturbed sites. The number of reptile species caught in restored sites were lower (9–16 species) than in adjacent, undisturbed sites (17–35 species). Reptile abundances were generally less on restored sites than undisturbed adjacent sites (results reported as model outputs). Five former mine site waste dumps, where restoration had started three–nine years previously, and an unmined area adjacent to each dump were monitored. At four mines, pitfall traps and drift fencing were used to survey sites over a seven-day period, on 10 occasions, from spring 2000 to winter 2002. At one mine, surveying was carried out five times, from spring 2001 to winter 2002.

A replicated, site comparison study in 1978–2005 of former mines (total number not given) in jarrah forests in Western Australia, Australia (4, same experimental set-up as 1) found that restored areas were recolonized by a range of reptile species. Of 24 reptile species commonly found in upland jarrah forest, 21 were recorded in restored forest sites. The following results were not statistically tested. Two generalist reptile species (skink *Tiliqua rugosa* and Dugite snake *Pseudonaja affinis*) were recorded in restored sites after 2–3 years. Authors report that more specialist species (e.g. *Phyllodactylus marmoratus* and *Ramphotyphlops australis*) have only been observed in restored sites >12 years old (personal observations of other sites). The authors reported that older-style restored areas (established using trees only) were unsuitable for most reptile species, whereas more advanced restoration approaches (including direct topsoil return) supported more species and greater abundance of reptiles. Some reptile abundances were lower in restored forest compared to unmined areas (see original paper for details). In 1976, two sites were restored by seeding (1 site) or top soil addition (1 site). In 1990, further sites (number not given) were restored using various techniques, including topsoil return, deep ripping, understorey seeding of many local species and establishment of local eucalypt species. Wildlife corridors and specific microhabitats (e.g. hollow logs, stumps) were created. Reptiles in restored areas (of varying ages and restoration techniques) and undisturbed forest were monitored in 1981, 1987, 1993, and 1999.

A before-and-after study in 2009 of a coal spoil prairie with wetlands in Indiana, USA (5, same experimental set-up as 11) found that restored areas were recolonized by snakes, turtles and one lizard species over 27 years. In total, 14 snake species (1–7 individuals encountered/species, Shannon-Wiener diversity index: 9), five turtle species (2–108 individuals encountered/species, Shannon-Wiener diversity index: 3) and one lizard species (5 individuals encountered/species) were recorded. Two were species of conservation concern: Kirtland's snake *Clonophis kirtlandii* and Eastern box turtle *Terrapene carolina*. Three snake species were new county records. In 1982–1983, an ex-mining area was graded to the approximate original contours, topsoil was added (15–38 cm) and the area was re-vegetated. Planting was initially of non-native tall fescue *Festuca arundinacea*, but since 1999 was replaced with native prairie grasses and forbs. Drift-fences with pitfall traps were installed (920 m) around four seasonal or semi-permanent wetlands and were sampled daily in March–August 2009. Visual encounters were also recorded.

A replicated, controlled study in 2002–2006 of forest at a restored mining site in Western Australia, Australia (6) found that thinning trees and burning vegetation as part of post-mining restoration increased reptile abundance and species richness. The effects of thinning and burning cannot be separated. Reptile abundance and richness in restored mining plots that were thinned and burned (abundance: 6.5–8.0 individuals/grid, richness: 3.8 species/grid) was higher than in plots that were not thinned and burned (abundance: 4.0–4.7 individuals/grid, richness: 1.5–1.7 species/grid). See paper for details of individual species responses. In 1984–1992, areas of a former bauxite mine were either planted with non-local tree species or sown with the seed of local tree species. Eight plots were thinned between December 2002 and July 2003 and then burned in November 2003. Eight different plots were not thinned or burned. Reptiles were monitored for four nights each in October and November–December 2005 and March and May 2006, using pitfall traps with drift fencing and live cage and box traps.

A replicated, controlled study in 2008–2009 in eucalypt forest in Western Australia, Australia (7) found that all wild Napoleon's skinks *Egernia napoleonis* reintroduced to restored bauxite mine sites moved to unmined forest within a week of being released and travelled further each day than skinks released directly into unmined forest. Six of six Napoleon's skinks translocated to restored mine sites moved into unmined forest within 7 days. In the first 30 days, skinks released into restored mine sites travelled greater daily distances (4.0 m/day) compared to skinks released into unmined forest (1.9 m/day). All 12 skinks (6 released in restored mining sites; 6 released in unmined sites) had settled in unmined forest after four months, but skinks released into restored mine sites still travelled more each day (3.0 m/day) than skinks originally released into unmined forest (0.4 m/day). In November 2008, twelve Napoleon's skinks were released in three five-year-old restored forest sites and three unmined forest sites (two skinks/site; see original paper for details of restoration). Skinks were radio-tracked weekly for the first four weeks after release and then monthly for the next three months. Skinks were recaptured and weighed monthly.

A replicated, site comparison study in 2005–2006 in two eucalypt forest sites in Western Australia, Australia (8) found that restored ex-mining forest reptile communities were different to unmined forest after 4–17 years. Up to 17 years after eucalypt forest restoration in two mine pits, reptile communities were different in restored forest compared to unmined forest (data reported as statistical model outputs, see original paper for details). Of the most commonly caught species, five species were observed in both restored and unmined forest, of which three species were less abundant in restored than unmined forest, and two species were less abundant in 8–17-year-old restored forest compared to four-year-old restored forest or unmined forest (see paper for details on individual species abundances). Reptiles were monitored in two restored mine pits in areas with four, eight, 12 and 17-year-old restoration forest plantings and unmined forest (six plots/forest type, split between the two mine pits). Details on forest restoration are not provided. Reptiles were surveyed using drift fences, and funnel and pitfall traps in October 2005 and January, March and May 2006 (1,728 trap nights/plot type, 8,640 total trap nights). In total 20 reptile species were recorded (270 individuals).

A review of studies investigating the success of habitat restoration following cessation of mining activities in Australia (9) found that reptile species richness and densities tended to be lower in restored compared to undisturbed areas, and a range of other responses were measured in three studies or less. Restored sites tended to be worse than undisturbed sites when measuring reptile density (lower in restored compared to undisturbed sites in 6 of 9 studies) and species richness (lower in 11 of 12 studies). Evenness (worse in 1 of 1 studies), community composition (worse in 2 of 2 studies), diversity (lower in 1 of 2 studies), body size or condition (similar in 3 of 3 studies), behaviour change (differences found in 1 of 1 studies) and biomass (higher in 1 of 1 study) were also assessed in 1–3 studies each. One study also reported breeding in a restored site. Fourteen studies that measured 33 outcomes for reptiles were included in the review. Restored sites included in the review were formerly bauxite, sand, uranium, coal, gold, manganese or iron mines.

A study in 2009–2011 in tropical rainforest on Reunion Island, Indian Ocean (10) found that some artificial egg-laying sites in a restored area in a hydroelectric power plant were used by Reunion day geckos *Phelsuma borbonica* in the year they were installed and the number of sites used and eggs laid increased in the second year. The effects of restoration and the provision of egg laying sites cannot be separated. Nine months after artificial egg-laying sites were installed, four of 34 sites were used by geckos and 10 eggs had been laid. Two years after the first artificial egg-laying sites were installed, eight of 40 sites were used by geckos and 41 eggs had been laid. Native plants (22,000 plants of 50 species) were planted in an area (9,000 m^2) of degraded habitat in a hydroelectric power plant to restore habitat. In September 2009–July 2010, forty artificial egg-laying sites were added to one area (34 were installed by June 2010 and a further six by July 2010). Artificial egg-laying sites comprised hollow, rectangular metal poles (4 x 8 x 250 cm) inserted into the ground (50 cm deep). Egg-laying sites were monitored for signs of geckos and egg laying in June and September 2010, and March and September 2011.

A replicated study in 2009–2010 in two ephemeral ponds in Indiana, USA (11, same experimental set-up as 5) found that snakes and turtles colonized a restored former open cast coal mine within 30 years. Following reseeding and restoration of a former open cast coal mine, four turtle species (10–198 individuals/species) and seven snake species (1–16 individuals/species) colonised two ephemeral ponds within 30 years. Between 1976 and 1982, the study site (729 ha) was a 30 m deep, open pit strip mine. Following mine closure, in 1982 the area was contoured and seeded to herbaceous cover vegetation initially and then in 1988 re-seeded to prairie grass species. As a result of mining activities, the area contained several waterbodies, including two ephemeral pond and wetland areas (0.14–0.33 ha). These wetlands were surveyed in March–October 2009 and March–August 2010 using drift fences and pitfall traps around the ponds (270–280 m of fencing/pond and 26–27 pitfall traps/pond).

A replicated, site comparison study in 2005–2012 in eucalypt forest in southwestern Australia (12) found that restored ex-mined forest did not maintain the same reptile assemblages as unmined forest up to 20 years after mining ceased. Reptile assemblages in restored ex-mining sites of all ages were different to unmined sites and did not become more similar to unmined sites over time (all results reported as statistical model outputs, see original paper for details). All 17 reptile species found in unmined sites were also found in restored sites, but 10 of 17 species were less abundant in restored sites. See original paper for details of individual species abundance changes over time. After bauxite mining ceased, eucalypt forest patches (~20 ha each) were restored by replacing retained topsoil and re-establishing vegetation from the topsoil seedbank, direct seeding and planting. In 2005–2012, reptiles were surveyed in 104 ex-mining sites that were restored 3–20 years earlier and 35 unmined sites. Reptiles were trapped using drift fences with pitfall and funnel traps in October–December and March (restored sites: 25,920 trap nights, unmined sites: 9,216 trap nights; trapping did not occur every year).

A replicated, site comparison study in 1990–1992, 2005–2006 and 2010–2011 in eucalypt forest in Western Australia, Australia (13) found that 20–22 year-old restored mining sites that were thinned and burned had similar species richness and abundance to restored sites that were not thinned and burned, but restored forest overall had lower species richness compared to unmined forest. Seven years after burning and tree thinning (management) took place, reptile species richness was similar between managed-restored forest (5 species/plot) and unmanaged-restored forest (4 species/plot) but richness in both restored forest types was lower than in unmined forest (9 species/plot). Reptile abundance was similar in managed-restored (21 individuals/plot) and unmined forest (34 individuals/plot). Abundance in unmanaged-restored forest (10 individuals/plot) was lower than in unmined forest, but similar to managed restored forest. See original paper for details of individual reptile abundances. The study area was restored after mining in 1990–1992 by reseeding with local vegetation. Reptiles were surveyed in four plots in each of managed-restored forest, unmanaged-restored forest and unmined forest. Managed-restored forest was thinned by felling (December 2002–June 2003) and prescribed burning (November 2003, reduced to 600–800 stems/ha) and two plots were re-thinned in January–December 2009 (reduced to 400 stems/ha). Unmined forest was prescribed burned 3–5 years before surveys. Reptiles were monitored using drift fences with funnel and pitfall traps in 2005–2006, 2010, and 2011.

(1) Nichols O.G. & Bamford M.J. (1985) Reptile and frog utilisation of rehabilitated bauxite minesites and dieback-affected sites in Western Australia's Jarrah *Eucalyptus marginata* forest. *Biological Conservation*, 34, 227–249.

(2) Galán P. (1997) Colonization of spoil benches of an opencast lignite mine in northwest Spain by amphibians and reptiles. *Biological Conservation*, 79, 187–195.

(3) Thompson, G.G. & Thompson S.A. (2005) Mammals or reptiles, as surveyed by pit-traps, as bio-indicators of rehabilitation success for mine sites in the Goldfields region of Western Australia? *Pacific Conservation Biology*, 11, 268–286.

(4) Nichols O.G. & Grant C.D. (2007) Vertebrate fauna recolonization of restored bauxite mines - key findings from almost 30 years of monitoring and research. *Restoration Ecology*, 15, S116–S126.

(5) Lannoo M.J., Kinney V.C., Heemeyer J.L., Engbrecht N.J., Gallant A.L. & Klaver R.W. (2009) Mine spoil prairies expand critical habitat for endangered and threatened amphibian and reptile species. *Diversity*, 1, 118–132.

(6) Craig M.D., Hobbs R.J., Grigg A.H., Garkaklis M.J., Grant C.D., Fleming P.A. & Hardy G.E.S.J. (2010) Do thinning and burning sites revegetated after bauxite mining improve habitat for terrestrial vertebrates? *Restoration Ecology*, 18, 300–310.

(7) Christie K., Craig M.D., Stokes V.L. & Hobbs R.J. (2011) Movement patterns by *Egernia napoleonis* following reintroduction into restored jarrah forest. *Wildlife Research*, 38, 475–481.

(8) Craig M.D., Hardy G.E.S.J., Fontaine J.B., Garkakalis M.J., Grigg A.H., Grant C.D., Fleming P.A. & Hobbs R.J (2012) Identifying unidirectional and dynamic habitat filters to faunal recolonisation in restored mine-pits. *Journal of Applied Ecology*, 49, 919–928.

(9) Cristescu R.H., Frère C. & Banks P.B. (2012) A review of fauna in mine rehabilitation in Australia: Current state and future directions. *Biological Conservation*, 149, 60–72.

(10) Sanchez M. (2012) Mitigating habitat loss by artificial egg laying sites for Reunion day gecko *Phelsuma borbonica*, Sainte Rose, Reunion Island. *Conservation Evidence*, 9, 17–22.

(11) Terrell V.C.K., Klemish J.L., Engbrecht N.J., May J.A., Lannoo P.J., Stiles R.M. & Lannoo M.J. (2014) Amphibian and reptile colonisation of reclaimed coal spoil grasslands. *Journal of North American Herpetology*, 1, 59–68.

(12) Triska M.D., Craig M.D., Stokes V.L., Pech R.P. & Hobbs R.J. (2016) The relative influence of in situ and neighborhood factors on reptile recolonization in post-mining restoration sites. *Restoration Ecology*, 24, 517–527.
(13) Craig M.D., Smith M.E., Stokes V.L., Hardy G.E.S.T.J. & Hobbs R.J. (2018) Temporal longevity of unidirectional and dynamic filters to faunal recolonization in post-mining forest restoration. *Austral Ecology*, 43, 973–988.

4.5. Use fencing to prevent reptiles from accessing facilities

- We found no studies that evaluated the effects on reptile populations of using fencing to prevent reptiles from accessing facilities.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

There is a risk to reptiles from becoming trapped, falling into mining pits or being electrocuted in mining or energy production facilities. Fencing may be used around such sites to deter reptile entry. As well as reducing direct risks to reptiles, if successful the intervention may also reduce the need to carry out lethal control of reptiles on such sites.

5. Threat: Transportation and service corridors

Background

The greatest threats from transportation and service corridors tend to be from the destruction of habitat and pollution. Interventions in response to these threats are described in the chapters *Habitat restoration and creation* and *Threat: Pollution* and actions to protect reptiles by moving them away from construction zones are described in the chapter *Species management* under *Mitigation translocations.* However, often a more visible impact is the mortality of reptiles in collisions with road vehicles. Roads or railways may also present physical barriers that prevent reptiles accessing suitable patches of habitat or disrupt daily or seasonal movements (van der Ree *et al.*, 2015). The same may be true for pipelines. Substantial efforts can be put into reducing these threats, through actions such as providing underpasses or overpasses. Monitoring frequently just considers use of these structures rather than the overall effect on population status of target species. Interventions mitigating negative impacts of boat traffic on aquatic reptiles are also considered in this chapter under *Aquatic transport corridors and boats.*

Van Der Ree R., Smith D.J. & Grilo, C. (2015) *Handbook of road ecology*. John Wiley & Sons, Ltd, UK.

Terrestrial Roads, Railroads & Service Corridors

5.1. Install barriers along roads/railways

- **Seven studies** evaluated the effects of installing barriers along roads/railways on reptile populations. Six studies were in the USA[1-6] and one was in Canada[7].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (3 STUDIES)

- **Survival (3 studies):** One before-and-after study in the USA[4] found that following installation of a barrier fence, along with creating artificial nest mounds on the non-road side of the fence, and actively moving turtles off the road, fewer turtles[4] were found dead on the road. One before-and-after study in the USA[6] found that following installation of a roadside barrier with nest boxes along with a warning sign, fewer female diamondback terrapins were killed while crossing the road compared to before installation. One study in Canada[7] found that dead snakes were found in the vicinity of a barrier fence up to 11 years after it was installed.

BEHAVIOUR (4 STUDIES)

- **Use (4 studies):** One controlled, before-and-after study in the USA[5] found that following installation of a roadside barrier with nest boxes, fewer diamond-backed terrapin crossed the road compared to before installation. One replicated study in the USA[3] found that after installing barriers, diamondback terrapins laid more nests on the marsh-side of the fence than on the road-side. The study[3] also found that terrapins were less likely to breach barriers with smaller gaps at the bottom. One replicated study in the USA[1] found that desert tortoises were effectively blocked by a concrete barrier. One replicated study in the USA[2] found that taller fences were better at excluding painted and snapping turtles than lower ones.

- **Behaviour change (1 study):** One replicated study in the USA[1] found that desert tortoises interacted less with solid compared to non-solid barriers.

Background

Wildlife barriers aim to prevent animals from crossing roads. They are typically wire mesh fences running parallel to the road. Although fencing may protect wildlife from traffic, it should not create an absolute barrier that prevents migration, isolates populations, fragments habitat, causes injuries, or traps reptiles (e.g. Barton & Kinkead 2005, Wilson & Topham 2009, Kapfer & Paloski 2011, Ferronato *et al*. 2014). Wildlife fencing is therefore usually combined with safe crossing opportunities such as wildlife underpasses and overpasses (see *Install overpasses over roads/railways* and *Install tunnels/culverts/underpasses under roads/railways*).

Studies included here are those that specifically assess barrier effectiveness, sometimes in combination with other collision reduction actions, but not where effects of fencing cannot be separated from effects of road underpasses. For these interventions combined, see *Install barriers and crossing structures along roads/railways*.

Barton C. & Kinkead K. (2005) Do erosion control and snakes mesh? *Journal of Soil and Water Conservation*, 60, 33A–35A.

Wilson J.S. & Topham S.E.T.H. (2009) The negative effects of barrier fencing on the desert tortoise (*Gopherus agassizii*) and non-target species: is there room for improvement. *Contemporary Herpetology*, 3, 1–4.

Kapfer J.M. & Paloski R.A. (2011) On the threat to snakes of mesh deployed for erosion control and wildlife exclusion. *Herpetological Conservation and Biology*, 6, 1–9.

Ferronato B.O., Roe J.H. & Georges A. (2014) Reptile bycatch in a pest-exclusion fence established for wildlife reintroductions. *Journal for Nature Conservation*, 22, 577–585.

A replicated study in 1991–1992 in an outdoor facility and along a highway in Nevada, USA (1) found that desert tortoises *Gopherus agassizii* interacted less with solid than non-solid barriers and that a concrete barrier was effective in keeping tortoises from a road. In daytime, desert tortoises spent less time close to and touched or pushed solid barriers less often (4–19 minutes, 0.4–4 touches/trial, 0.1 pushes/trial) than non-solid mesh barriers (5–23 minutes, 2–12 touches/trial, 5 pushes/trial) and in shorter trials made less attempts to climb over solid than non-solid fences (30 minute trial: 1 attempts/trial vs. 2 attempt/trial, 2 h trial: 0.1 vs. 0.1, night time: 1 vs. 0.1). The authors reported tortoises frequently attempted to walk through fences they could fit their head through. In a separate maze experiment, tortoises showed no preference for solid or mesh fencing (see paper for details). In a trial by a highway, nine of 10 tortoises approached a concrete barrier and walked along it for an average of 13 m before seven walked away from the highway (the remaining two tortoises settled in place). Tortoises were placed individually in pens with solid (e.g. cabin timber, aluminium flashing, cement blocks, telephone poles) or non-solid walls (e.g. chain link, chicken wire, mesh cloth) for 30 minute (solid: 41 trials, non-solid: 22), 2 hour (160, 100 trials) or overnight trials (40, 80 trials) and behaviours monitored. Tortoises participated in one trial/wall material and three trials maximum each. In separate trials, 16 tortoises were placed in a T-shaped maze with a choice between navigating towards solid or mesh fencing (40 total trials) and 10 tortoises were placed by a concrete barrier next to a highway and observed. All trials took place in 1991–1992.

A replicated study in 2005–2006 in a Wildlife Management Area, in New York, USA (2) found that plastic fences of at least 0.6 m high excluded all painted *Chrysemys picta* and snapping turtles *Chelydra serpentina.* Fences 0.6 m and 0.9 m high were more effective at excluding turtles (100% excluded) than 0.3 m high fences (84–100%). Opaque, corrugated plastic fences were used to construct three nested, circular enclosures of heights 0.3, 0.6 and 0.9 m high. Local painted (74 individuals) and snapping turtles (62 individuals) were placed in the centre of each arena and left for 15 minutes to attempt to scale the fences.

A replicated study in 2011–2012 along two roadside verges across salt marshes in New Jersey, USA (3) found that where barriers were installed, diamondback terrapins *Malaclemys terrapin* nested more often on the marsh-side of barriers than on the road-side, and that terrapins were less likely to breach barriers with smaller gaps at the bottom. After barrier fences were installed, diamondback terrapins laid more nests on the marsh-side of the fence than on the road-side (results presented as statistical model outputs, see original paper for details). In separate arena trials, diamondback terrapins were less likely to breach fences with smaller gaps at the bottom (0 cm gap: 10% breached; 3–5 cm: 37–60 %; 6–8 cm: 96–100%). In 2011 and 2012, sections of two causeways (589–623 m long) with corrugated tubing fencing (15 cm diameter) were surveyed on foot for terrapin nests. Surveys were carried out every week in June–July 2011 and twice in June–July 2012. Trials to test whether terrapins could breach the fences with different sized gaps at the bottom (0–8 cm) were carried out in a fenced enclosure with 40 individual terrapins (total of 74 trial).

A before-and-after study in 1999–2008 on a roadside verge along a river bank in Pennsylvania, USA (4) found that after adding a chain-link fence to a highway, creating artificial nest mounds on the non-road side of the fence, and actively moving turtles off the road, fewer female northern map turtles *Graptemys geographica* were killed on the road. Results were not statistically tested. In the first year after a fence was installed on a new major highway, 10 northern map turtles were killed on the road, compared to 50 turtles the year before (total for previous year included a small number of wood turtles *Glyptemus insculpta* and snapping turtles *Chelydra serpentina*). In the subsequent 8 years, map turtle deaths reduced to 0–5 individuals/year (no data for other species). The authors reported that most deaths were gravid female turtles. In 2000, a fence (1 m high and 1,150 m long) was installed on the river side of the highway to prevent turtles from crossing the road to access nesting habitat. Mounds of sand aimed at providing alternative nesting habitat were added to the river side of the fence in 2000–2001. After the first year, the fence was extended by 300 m to prevent turtles from going around it and crushed shale was added to the sand mounds and turtles were actively moved off the road. Turtle deaths on the road were counted from 1999 (the first year after a new highway opened) to 2008 (excluding 2004).

A controlled, before-and-after study in 2009–2014 on a causeway over a saltmarsh in Georgia, USA (5, same experimental set-up as 6) found that installing a roadside barrier with nest boxes reduced diamond-backed terrapin *Malaclemys terrapin* road crossings. Numbers of crossings by terrapins were lower at the site after the barrier was introduced (2–7 crossings) compared to before (10–17 crossings) and compared to sites without the barrier (12–109 crossings). Three monitoring locations along the causeway were selected based on high terrapin road mortality levels in 2009–2010. Two sites (331 m and 310 m long stretches of road) without barriers were monitored, and at a third site (162 m long, between the other sites) a barrier was installed in 2011. The barrier was 22.9 m long, positioned 5 m from the roadside and comprised short mesh fencing and six nest boxes. Terrapins were monitored on the road at all three sites every 20–90 minutes, between 08:00 and 20:00 from May–July in 2009–2014. Two years of pre-barrier and four years of post-barrier data were collected.

A before-and-after study in 2009–2015 on a causeway over a saltmarsh in Georgia, USA (6, same experimental set-up as 5) found that installing a roadside barrier with nest boxes and adding a flashing terrapin-warning sign to alert motorists reduced likelihood of mortality for diamondback terrapin *Malaclemys terrapin* crossing the road. When the hybrid nestbox-fence barrier and flashing signs were added to a road, survival of crossing female diamondback terrapins increased from 24% to 53% (data reported as statistical model outputs). In 2011, a 22 m long hybrid nestbox-fence barrier was built along an 8.7 km long causeway. In 2013, two terrapin crossing signs with flashing warning beacons were added to warn motorists entering a 6 km stretch of causeway. The signs were activated for 2 hours/day during peak terrapin crossing times. Terrapins were surveyed on the causeway and in adjacent creeks during the nesting season (May–July) in 2009–2015.

A study in 2006–2017 in shrub-steppe desert in British Columbia, Canada (7) found that building an exclusion fence to prevent snakes entering high human areas activity and associated roads did not prevent road related snake mortality. In the first year after the exclusion fence was installed, seven snake mortalities were observed near the fence. Ten to 11 years after the fence was installed, 22 of 45 (49%) snake deaths were road kill, and a further 15 (33%) dead snakes were found next to the fence. Six of the 15 dead snakes near the fence were found next to a section that had been rerouted the previous year. Western yellow-bellied racer *Coluber constrictor mormon* were disproportionately represented among dead snakes. In total 341 live snakes (northern pacific rattlesnake *Crotalus oreganus oreganus*, great basin gopher snake *Pituophis catenifer deserticola*, and western yellow-bellied racer) were captured around the fence in the tenth and eleventh year after it was installed. In 2006, approximately 4 km of exclusion fencing (60 cm high with 0.6 cm mesh) was built. The fence was upgraded and 200 m rerouted in 2015. Snake mortality was monitored in May–October 2016–2017 by walking the fence line 2–3 times/week and live snakes were monitored using mark-recapture methods 5–6 days/week.

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(2) Woltz H.W., Gibbs J.P. & Ducey P.K. (2008) Road crossing structures for amphibians and reptiles: informing design through behavioral analysis. *Biological Conservation*, 141, 2745–2750.
(3) Reses H.E., Davis Rabosky A.R. & Wood R.C. (2015) Nesting success and barrier breaching: Assessing the effectiveness of roadway fencing in diamondback terrapins (*Malaclemys terrapin*). *Herpetological Conservation and Biology*, 10, 161–179.
(4) Nagle R.D. & Congdon J.D. (2016) Reproductive ecology of *Graptemys geographica* of the Juniata river in Central Pennsylvania, with recommendations for conservation. *Herpetological Conservation and Biology*, 11, 232–243.
(5) Crawford B.A., Moore C.T., Norton T.M. & Maerz J.C. (2017) Mitigating road mortality of diamond-backed terrapins (*Malaclemys terrapin*) with hybrid barriers at crossing hot spots. *Herpetological Conservation and Biology*, 12, 202–211.
(6) Crawford B.A., Moore C.T., Norton T.M. & Maerz J.C. (2018) Integrated analysis for population estimation, management impact evaluation, and decision-making for a declining species. *Biological Conservation*, 222, 33–43.
(7) Eye D.M., Maida J.R., McKibbin O.M., Larsen K.W. & Bishop C.A. (2018) Snake mortality and cover board effectiveness along exclusion fencing in British Columbia, Canada. *Canadian Field-Naturalist*, 132, 30–35.

5.2. Install barriers and crossing structures along roads/railways

- **Sixteen studies** evaluated the effects of installing barriers and crossing structures along roads/railways on reptile populations. Five studies were in the USA[2,5-7,11], three were in each of Spain[1,9,10], Australia [4,8,13] and Canada[12,14,16] and one was in each of France[3] and South Africa[15].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (8 STUDIES)

- **Survival (8 studies):** Four of seven studies (including one randomized, controlled, before-and-after study and one review) in the USA[5-7,11], Australia[8], Canada[12] and South Africa[15] found that installing fencing and crossing structures did not reduce road mortalities of reptiles[6,8,12,15], and in one case the percentage of mortalities may have increased[12]. Two studies[7,11] found that areas with fencing and crossing structures had fewer road mortalities of turtles[7] and overall reptiles[11]. One study[5] found that reptile road mortalities still occurred in in areas with roadside barrier walls and culverts. One replicated, before-and-after study in Canada[14] found that following installation of tunnels and guide fencing, along with signs for motorists, there were fewer road mortalities of eastern massasauga rattlesnakes.

BEHAVIOUR (12 STUDIES)

- **Use (12 studies):** Six studies (including two replicated studies and one review) in Spain[1,9], France[3], the USA[6,7]and Australia[13] found that crossing structures with fencing that were not specifically designed for wildlife were used by lizards[1,9,13], snakes[1,6,13], tortoises[3,6,], turtles and alligators[7] and ophidians[9]. One study[1] also found that the addition of fencing around crossing structures did not affect the number of reptile crossings. Three studies (including one replicated and one before-and-after study and one review) in the USA[2,6] and Spain[9] found that wildlife crossing structures with fencing were used by gopher tortoises and 12 snake species[2], American alligators[6] and lacertid lizards[9]. One study[2] also found that an American alligator did not use the wildlife crossing structure. Two before-and-after studies (including one controlled study) in Canada[12,16] found mixed effects of installing roadside fencing and culverts on road use by turtles and snakes. One replicated study in Spain[10] found that use of different crossing structures depended on species group. One replicated study in Australia[4] found that reptiles used wildlife underpasses or culverts for only 1% of road crossings. One replicated, before-and-after study in Canada[14] found that following installation of tunnels and guide fencing, along with signs for motorists, fewer eastern massasauga rattlesnakes were found crossing the road.

Background

Schemes designed to reduce collisions between vehicles and reptiles may use multiple interventions. Two of the most common ones, installing barrier fencing and providing routes for reptiles to travel underneath or over roads, are often employed within the same scheme. This may entail regular roadside fencing with entrances to underpasses set further back away from the road or fencing may be designed to adjoin the sides of under or overpass entrances. Sometimes, fencing may be installed to form a funnel leading towards under or overpass entrances.

This intervention includes studies where these two actions are in place at the same site. In most studies, all under or overpasses (where there are multiple crossings) are beneath stretches or roads that have barrier fencing. In a minority, just some of the under or overpasses monitored are along stretches with barrier fencing. Studies included use of either conventional fencing, electric fences or other barriers, such as walls. Most studies report solely on the use of crossings or trends in numbers of reptiles killed on roads. There is an absence of studies reporting on wider population-level effects of the presence of these structures.

See *Install tunnels/culverts/underpasses under roads/railways* or *Install overpasses over roads/railways* for studies where under or overpasses are either installed without use of barriers or where it is not clear from the study that barriers were installed. See also *Install barriers along roads/railways* for studies where the specific effect of barriers was evaluated.

A study in 1991–1992 along a high-speed railway through agricultural land in Castilla La Mancha, Spain (1) found that culverts and underpasses not specifically designed for wildlife were used as crossings under the railway by reptiles, but that the addition of fencing did not affect crossing rates. Lizards and snakes were recorded making 112 crossings, or 7 crossings/100 passage-days on average across 15 underpasses and two overpasses. Reptiles preferred culverts 2 m wide and used culverts or underpasses more frequently than overpasses. Fencing did not significantly affect relative crossing rates (data presented as statistical model result). Reptile crossing rates were lower in autumn–early spring and varied with habitat types. Fifteen dry culverts and passages (e.g. small roads and two overpasses, 13–64 m long, 1.2–6.0 m wide, 1.2–3.5 m high) along a 25 km section of high-speed railway, were monitored. The railway was fenced with 2 m high wire netting in July 1991–March 1992. Tracks in sand were monitored at each passage for 15–22 days/month between September 1991 and July 1992.

A before-and-after study in 1993–1995 in forest and pasture in Florida, USA (2) found that after a fence and wildlife underpass was built, numbers of road-killed reptiles did not decrease, but tortoises and snakes used the crossing. Sample sizes were small and results were not statistically tested. In the year after a fenced wildlife underpass was installed, two box turtles *Emydidae* spp., one cooter *Pseudemys concinna* and one gopher tortoise *Gopherus polyphemus* were killed on the road, compared to one box turtle before installation. The underpass was used by two gopher tortoises and 12 snakes (species not identified). One alligator *mississippiensis* walked along the fence line but did not use the crossing. A wildlife underpass (14.3 m long, 7.3 m wide, 2.4 m high) and barrier fencing (3 m chain-link and barbed wire fence topped with three strands of barbed wire, both sides of the underpass, 1.7 km total length) was erected in 1994. Trails were cut into woodland and trees planted in pasture to guide wildlife to the underpass. Roadkill surveys were carried out three times/week pre-fencing (December 1993–November 1994) and post-fencing (December 1994–November 1995). In December 1994–December 1995, movements along the fence line and in the underpass were monitored by surveying tracks and a camera in the centre of the underpass.

A study in 1989–1994 of roadside verges in Toulon, France (3) found that some Hermann's tortoise *Testudo hermannii* used culverts or a road tunnel with fencing to cross a highway. Seven (three females, four males) of 70 individually-marked Hermann's tortoises used a road tunnel or culverts to cross a highway. A highway was constructed between May 1989 and October 1990 through Hermann's tortoise habitat. Sheep wire fencing was erected to prevent tortoises from crossing the road and one road tunnel and two culverts were constructed to allow tortoise movements between the two sides of the highway. Resident tortoises (300 individuals) were temporarily relocated during construction and individually marked prior to release. In April–October 1993–1994, observers carried out visual searches for tortoises next to the highway, recording recaptures (70 relocated tortoises were recaptured) and individually marking new individuals.

A replicated study in 2000 in nine roadside verges in coastal open woodland in New South Wales, Australia (4) found that wildlife underpasses ('culverts') with fencing were used by some lizards and snakes in a four-month period. Road underpasses were used 11 times by lizards, including three lace monitor *Varanus varius* crossings (the only species mentioned) and twice by snakes. Reptile use of the underpasses comprised 1% of all wildlife crossings (1,202 total crossings). Nine purpose-built wildlife underpasses were surveyed for wildlife crossings along a 1.4 km long section of road traversing coastal low-lying dunes. Both sides of the road were lined with a 180 cm high chain-mesh fence. Culverts were made from reinforced concrete with stone or silt floors. Reptiles were surveyed using sand strips across the middle of each culvert (1 m long, 2–3 cm deep). Sand was checked for tracks every second day for eight days in September 2000 and December 2000.

A study in 2001–2002 along a highway in Florida, USA (5) found that culverts, in areas with roadside barrier walls, were used by reptiles but road casualties still occurred. Seventeen reptile species were recorded using culverts. These included American alligators *Alligator mississippiensis* (in five culverts), four turtle species (four culverts), green anoles *Anolis carolinensis* (one culvert) and 11 snake species (seven culverts). During the same period, ≥22 reptile species were recorded dead on the road. The most frequent casualties were yellow ratsnake *Elaphe obsolete* (16 individuals), southern watersnake *Nerodia fasciata* (21), and DeKay's brownsnake *Storeria dekayi* (54). Culverts reduced overall vertebrate road mortality, but separate reptile figures were not reported for before culverts were installed. Eight culverts (from 0.9 m diameter to 2.4 × 2.4 m cross-section, all 44 m long) were connected using prefabricated concrete barrier walls. Culverts were monitored from 14 March 2001 to 5 March 2002 using funnel traps, camera traps and sand track stations. Roadkills were monitored by walking the 3.2 km road over three consecutive days each week over the same period.

A review of studies investigating culverts and road barriers in the USA (6) found that some species used culverts and in some cases road casualties were reduced. Nine alligators *Alligator mississippiensis* used four fenced wildlife underpasses. Desert tortoise *Gopherus agassizii* and shovel-nosed snake *Chionactis occipitali* road casualties reduced after a barrier fence, 24 culverts and three bridges were installed and tortoises were recorded using the culverts. Although red-sided garter snakes *Thamnophis sirtalis infernalis* used tunnels, snake road casualties remained high during autumn migrations and only two timber rattlesnakes *Crotalus horridus* were recorded using a culvert in the two years following its construction. See original paper for details of each study.

A before-and-after study in 2000–2003 along a highway in Florida, USA (7) found that turtle road mortality decreased following the installation of drift fencing leading to a culvert. Turtle mortality on a road, primarily Florida cooter *Pseudemys floridana* and yellow-bellied slider *Trachemys scripta*, decreased after drift fencing was added to a culvert (0.1 individuals/km/day) compared to beforehand (11.9 individuals/km/day). During the study >200 individual turtle tracks and >25 alligator tracks were observed in the culvert. There was no evidence of predation of turtles at the culvert. In 2000, vinyl fences (600–700 m long, 0.6 m high) were installed along each side of a four-lane causeway to divert animals towards a culvert (diameter: 3.5 m, length: 46.6 m, built in 1963–1965). The bottom fence edge was buried approximately 20 cm underground and fence ends curved away from the road after >80–100 m. The highway and culvert were monitored using visual searches two–four times/day before (February–April 2000) and after (April 2000–November 2003) the construction of the fence (1,367 total survey days).

A before-and-after study in 2004–2007 in dry eucalypt woodland in Queensland, Australia (8) found that after an exclusion fence and vegetated overpass ('land-bridge') were built, one snake was found dead on the road compared to two before. Before construction of the fencing and overpass, one brown tree snake *Boiga irregularis* and one eastern small-eyed snake *Cryptophis nigrescens* were found as road-kill in the area and after the underpasses were constructed one carpet python *Morelia spilota* was found. In 2004 an exclusion fence, made of rubberised metal mesh (2.5 m high, 5 cm underground) with a rubber sheet (0.5 m high) running around the base, was built along a road overpass (see original paper for details) to a forest boundary on both sides of the bridge. Road-killed animals were surveyed by vehicle in the early mornings twice weekly before construction started in April–July 2004 and weekly after construction was completed in February 2005 until June 2007. All species larger than a blue-tongued skink *Tiliqua scincoides* were recorded.

A replicated study in 2002 of a highway in Zamora, Spain (9; same experimental set-up as 10) found that underpasses and culverts, in areas with roadside barrier fencing, were used by reptiles. Lacertid lizards (*Lacerta* spp. and *Podarcis* spp.) and ophidians (snakes and legless lacertids) were recorded in circular culverts (lacertids: 1.49 crossings/day/structure, ophidians: 0.03), adapted culverts (1, 0.5), and open-span underpasses (0.07, 0.07). Lacertid lizards were also recorded in wildlife underpasses (0.86 crossings/day/structure). A total of 64 underpasses and culverts (30–150 m long) under a 72 km section of motorway were monitored. These included 33 circular drainage culverts (2 m diameter), 10 wildlife-adapted box culverts (2–3 m wide, 2 m high), 14 open-span underpasses (rural tracks/paths, 4–9 m wide, 4–6 m high) and seven wildlife underpasses (20 m wide, 5–7 m high). The motorway was barrier-fenced. Animal tracks were monitored over 10 days in June–September 2002 using marble dust (1 m wide across).

A replicated study in 2001 along a highway in Zamora province, Spain (10; same experimental set-up as 9) found that road underpasses and culverts, in areas with roadside barrier fencing, were used by lizards. Lacertid lizards (*Lacerta* spp. and *Podarcis* spp.) were recorded in circular culverts (lacertids: 0.36 crossings/day/structure) and open-span underpasses (0.14) but not adapted culverts or wildlife underpasses. Ophidians (snakes and legless lacertids) were not recorded in any underpasses. Thirty-three crossings were monitored. These comprised 14 circular drainage culverts (2 m diameter, 35–62 m long), seven wildlife-adapted box culverts (2–4 m wide, 2–3 m high, 36–45 m long), seven open-span underpasses (rural tracks/paths, 4–9 m wide, 4–6 m high, 32–72 m long) and five wildlife underpasses (14–20 m wide, 5–8 m high, 30–96 m long). The motorway had barrier fencing along its length. Animal tracks were recorded using marble dust (1 m wide cross) over 10 days in March–June 2001.

A replicated, site comparison study in 2006–2007 on a highway through forest and agricultural land in North Carolina, USA (11) found that fenced stretches of road with underpasses tended to have lower rates of reptile road mortality than those without. Results were not statistically tested. Reptile mortality on stretches of road with underpasses and fencing was 1 reptile/km (8 individuals) compared to 2 reptiles/km (26 individuals) on unfenced road with no underpasses. Some reptiles, e.g. snakes, were small enough to climb through the fencing (see original paper). A new four-lane highway was constructed in 2001–2005 with three underpasses (3 m high, 29–47 m wide). Each underpass had an 800 m fence either side of it, which ran parallel to the highway, then continued under the underpass and connected with fencing on the opposite side (3 m high chain-link fencing). A section of the highway (with underpasses and fencing 6,375 m; without: 10,873 m) was surveyed for wildlife casualties twice/week in July 2006–July 2007.

A controlled, before-and-after study in 2012–2013 along a section of highway through wetlands, rocky outcrops and mixed forest in Ontario, Canada (12) found that installing fencing and culverts prevented an increase in road use by snakes, but may have increased the percentage of snakes and turtles that died on the road. The number of snakes and turtles (both dead and alive) discovered on roads stayed similar in areas with fencing and culverts (snakes: 0.6–0.7/day; turtles: 0.5/day), but without fencing and culverts snake numbers increased (before: 1.4/day; after: 2.4/day), but turtle numbers stayed the same (1.0–1.1/day).

However, the percentage of dead reptiles may have increased with fencing and culverts (before: 68% of turtles, 68% of snakes; after: 86% of turtles, 90% of snakes), but stayed similar in the area without (before: 86% of turtles, 76% of snakes; after: 88% of turtles, 88% of snakes), but this result was not tested statistically. In 2012, three crossing structures (culverts) were installed along a 13 km section of highway and connected with fencing (plastic sheeting and a chain-link fence). A similar 13 km section of highway with no fencing or culverts was also selected. In May–August in 2012 (before installation) and 2013 (after installation), both sections of highway were surveyed by car (13 km section; 3 surveys/day) or by foot (2 km section; 1 survey/day) to count the number of live and dead reptiles on the road.

A replicated study in 2012–2013 in southern metropolitan Perth, Western Australia, Australia (13) found that western bobtail lizards *Tiliqua rugosa* used underpasses of all sizes and shapes under fenced roads, and five other reptile species were observed using underpasses. Bobtail lizards used all 10 underpasses with 3–148 total crossings/underpass made by 1–8 individual lizards/underpass. Bobtail lizard use of underpasses was not related to length (23–88 m), cross-sectional area (0.3–1.4 m^2), presence of logs or sticks, surrounding vegetation cover (0–50%), presence of predators, or time since construction of the underpass (2–19 years). Other reptile species seen using the underpasses included Gould's sand monitor *Vaanus gouldii*, western bluetongue *Tiliqua occipitalis*, southern heath monitor *Varanus rosenbergi*, black-headed monitor *Varanus tristis* and dugite *Pseudonaja affinis*. Road crossings were monitored through 10 underpasses (round: 0.6–0.9 m diameter or square culverts: 0.6–1.2 m wide, 0.5–1.2 m high) from May 2012 to May 2013. All roads were fenced (600–1,800 mm high, buried 300 mm deep). Bobtail lizards were trapped and individually marked using PIT tags in the vicinity of each underpass for four consecutive nights. Underpasses were equipped with PIT tag reader antennas and infrared motion-sensor cameras installed to record animals on either side of the culvert.

A replicated, before-and-after study in 2002–2014 on three roads in Ontario, Canada (14) found that eastern massasauga rattlesnake *Sistrurus catenatus* mortality was reduced after tunnels and guide fencing, along with signs for motorists, were installed. The number of rattlesnakes found dead or alive on roads decreased after tunnel, fence and sign installation (dead: 6, alive: 14) compared to before installation (dead: 41, alive: 68) and during installation (dead: 15, alive: 37). Fourteen of 68 individually-marked ('PIT tagged') rattlesnakes were recorded using tunnels. Rattlesnakes and garter snakes *Thamnophis sirtalis* showed a similar willingness to enter tunnels (rattlesnakes: 18 of 19; garter snakes: 15 of 16) and complete crossings to the other side (rattlesnakes: 7 of 19; garter snakes: 5 of 16). In 2007–2013, three mesh barrier fences 600–900 m long were installed on one or two sides of the road). In 2010–2011, a grate-top tunnel (8.5 m long x 1.2 m wide x 0.5–0.6 m deep) was installed in two of the fenced sections. Four signs to encourage motorists to slow down for snakes were installed near known snake road crossing locations. Rattlesnakes were surveyed on the road in May–October before (2005–2007) and during installation (2008–2012) by car and after installation (2013–2014) by bicycle. Tunnel use was monitored with camera traps and automated PIT tag readers. In 2014, rattlesnakes and garter snakes were caught opportunistically and placed at tunnel entrances to test willingness to enter and complete tunnel crossings.

A randomized, controlled, before-and-after study in 2015 along a grassy verge in Limpopo Province, South Africa (15) found that adding trenches or barriers to direct reptiles to concrete road underpasses ('culverts') did not reduce reptile roadkill numbers. Overall numbers of small terrestrial vertebrates killed were statistically similar after trenches or barriers were erected (before: 0.33 roadkill/day/km vs after: 0.04 roadkill/day/km). After trenches or barriers were erected 0–1 reptiles were killed on the road compared to 1–2 individuals before they were put in place. Based on areas of high roadkill, pre-existing concrete culverts on a 12.3 km long road stretch were selected and randomly chosen to be adapted by the addition of a barrier (70 cm high, 30 cm below ground, 3 culverts), or a trench (30 cm deep, 3 culverts), or no changes were made (3 culverts). Trenches and barriers were built parallel to the road (approximately 2 m from the edge), 200 m long either side of the culvert (400 m total length). Roadkill surveys were carried out by vehicle for 20 consecutive days before trenches and barriers were built in January 2015 and afterwards in February 2015 (20 consecutive days).

A before-and-after study in 2003–2014 in a wetland complex in Ontario, Canada (16) found that adding roadside fencing and culverts reduced turtle and snake abundances on a causeway, although only along completely fenced sections of road, and use of culverts by individuals was low. In areas that were fully fenced, the number of turtle or snakes found on the causeway was lower after fencing than before (turtles: after fencing: 2, before fencing: 10; snakes: after: 3, before: 7), but remained similar in partially fenced areas (turtles: 3; snakes: 3–4) and areas with no fencing (turtles: 1–2; snakes: 2). Two of 68 Blanding's turtles *Emydoidea blandingii* and none of 30 spotted turtles *Clemmys guttata* used one of seven culverts. One of 30 radio-tracked Blanding's turtle used a culvert once. Reptiles were surveyed in April–October on a causeway (3.6 km long) across a marsh for five years (2003–2007: 22 surveys/month, 154 total surveys) before post-and-mesh-net fencing was installed in 2008–2009 and for five years afterwards (in 2010–2014: 40 surveys/month, 284 total surveys). After exclusion fencing was built, road sections were classified as: fully fenced, partially fenced, or unfenced. In 2012–2014, seven culverts were added to the causeway. In 2014–2015, culvert use was monitored by cameras, an automated PIT-tag checker at culvert entrances (68 Blanding's and 30 spotted turtles were PIT-tagged) and radio-tracking turtles (30 additional Blanding's turtles were radio-tracked once a week during active seasons).

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5.3. Install tunnels/culverts/underpasses under roads/railways

- **Fifteen studies** evaluated the effects of installing tunnels/culverts/underpasses under roads/railways on reptile populations. Four of the studies were in the USA[1,7,9,15], four were in Australia[4-6,8], three were in Spain[2,3,14], two were in Canada[11,12] and one was in each of Australia, Europe and North America[10] and South Africa[13].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (3 STUDIES)

- **Survival (3 studies):** Two site comparison studies (including one before-and-after study) in Australia[6] and South Africa[13] found a similar number of reptile road mortalities with or without culverts or wildlife underpasses. One replicated study in Spain[14] found that the number of underpasses in an area did not affect the number of reptile road mortalities.

BEHAVIOUR (12 STUDIES)

- **Use (12 studies):** Six studies (including four replicated studies and one replicated, before-and-after study) and one review in Spain[2,3], Australia[4,8], the USA[7,15] and Australia, Europe and North America[10] found that crossing structures, including tunnels, culverts, underpasses, pipes and trenches under roads and railways were used by reptiles[2,3,8,10], lizards[4], snakes[4,7] and/or tortoises[15]. One review in Australia, Europe and

North America[10] also found that wildlife underpasses were used by reptiles in only one of 13 studies. Three of four replicated studies (including one before-and-after study) in the USA[1,9] and Canada[11,12] found that desert tortoises[1], painted and snapping turtles[9] and rattlesnakes and garter snakes[12] showed a willingness to enter some, or all types of tunnel. The other study[11] found that only 9% of painted turtles entered a culvert during a choice experiment. One site comparison study in Australia[5] found that the area under an overpass was used by five reptile species.

Background

Tunnels, culverts and underpasses may provide safe crossing points for reptiles. A range of different tunnels can be used, including purpose-built wildlife tunnels, culverts that assist with drainage and which can also be used by wildlife, and large passages beneath elevated road sections which may sometimes also be used for local vehicle access. Culverts may be round (pipes) or square (box) and may or may not have natural substrate (sand or stones) on the bottom.

Underpasses are frequently installed in conjunction with wildlife barrier fencing which funnels animals towards the tunnel and prevents them from accessing the road or railway. For this combined intervention, see *Install barriers and crossing structures along roads/railways*. See also *Install barriers along roads/railways* and *Install overpasses over roads/railways*.

Studies included here are those where barrier fencing is not installed or not explicitly referred to in the study methods or where at least some underpasses were in unfenced areas. Most studies here report solely on the use of these structures, such as the number of crossings made. There is an absence of studies reporting on wider population-level effects of the presence of these structures.

A replicated study in 1992 in an outdoor facility in Nevada, USA (1) found that over half of desert tortoises *Gopherus agassizii* entered tunnels of a suitable size during trials, of which around half went all the way through tunnels. After 30 minutes, 12 of 16 tortoises had entered tunnels of a suitable width for their size and seven of the 12 tortoises escaped through or moved to the end of the tunnel. Two tortoises stopped in the tunnel and three tortoises entered the tunnel and returned to the pen. Four tortoises investigated tunnel openings but did not enter. In September 1992, sixteen tortoises were placed individually in the middle of a walled pen (4.6 x 4.6 m) with five tunnels of varying diameters and lengths located around the pen edge providing a choice of exit points (narrow: 10 cm wide x 150 cm long and 10 cm wide x 90 cm long; medium: 19 cm wide x 120 cm long; wide: 29 cm wide x 280 cm long and 29 cm wide x 136 cm long). Tortoise behaviour was observed for 30 minutes and recorded.

A replicated study in 1994 of 17 culverts under roads and railways in Madrid province, Spain (2) found that culverts were used by snakes and lizards. Lizards (0.23 tracks/day) and snakes (0.02 tracks/day) were detected. Snakes and lizards were detected in culverts more often in the summer (0.88 tracks/day) compared to spring (0.04 tracks/day), autumn (0.04 tracks/day) or winter (0 tracks/day). Culvert use by snakes and lizards declined when detritus pits were recorded near the culvert (see original paper for details). Five culverts were monitored under railways, two under a motorway and 10 under local roads. Structural, vegetation and traffic variables were recorded at each culvert. Use was monitored using marble (rock) dust over culvert floors to record tracks. Sampling was undertaken in 1994, over four days each in spring, summer, autumn and winter. Sampling extended to eight days at four culverts when deer were nearby.

A replicated study in 1993–1994 in four motorway roadside verges in Catalonia, Spain (3) found that road underpasses ('culverts') were used by reptiles and that reptiles were recorded more often in circular than rectangular underpasses. Results were not statistically tested. Reptiles used four of 17 rectangle-shaped underpasses and 23 of 39 circle-shaped underpasses. The authors reported that reptiles were only recorded crossing shorter-length underpasses and were more likely to be recorded when there was natural substrate on the floor of the underpass and the opening of the structure was at ground level. In November 1993–September 1994, fifty-six underpasses (including drainage channels) along four 10 km-long stretches of motorway were surveyed for use by reptiles. Thirty-nine underpasses were circular in cross section (1–3 m diameter) and 17 were rectangular (4–12 m wide). Each underpass was monitored four times for four days/season (16 days in total/underpass) using a 50 cm long strip of powder substrate across each underpass and infra-red and photographic cameras.

A replicated study in 1996–1997 along three roadside verges in New South Wales, Australia (4) found that road underpasses were used by three lizard species and one snake species. Over nine months, two of three underpasses were used by eastern water dragons *Physignathus lesueurii* (3 photographs, tracks observed in one underpass) and eastern water skinks *Eulamprus quoyii* (3 photographs, tracks observed in two underpasses). One of the underpasses was used by lace monitor *Varanus varius* (8 photographs, tracks and scats observed) and diamond python *Morelia spilota* ssp. *spilota* (one photograph). In mid-August 1996 to mid-June 1997, a camera with an infrared trigger was set in three different underpasses of different sizes and design on a highway. A 1 m wide sand tray was also placed in each underpass. The authors note that animals that were small enough to avoid triggering the camera may not have been consistently recorded.

A site comparison study in 2002–2003 in eucalypt forest in Victoria, Australia (5) found that 4–5 years after a flyover was built to enable wildlife to cross underneath a dual carriageway, some reptiles were present underneath the flyover as well as in adjacent forest. Four–five years after a road flyover was built, five species of reptile were counted underneath the flyover and five species of reptile were counted in forest adjacent to the flyover. In 1998 a dual carriageway road flyover was built across a tract of forest. Mature eucalypts and middle and understorey vegetation were kept during construction and native plant species were planted to maintain a similar vegetation structure to adjacent forest. Reptile use of the flyover was monitored monthly in July 2002–June 2003 using 14 different methods including pitfall trapping, sand trays and visual surveys for roadkill (see original paper for details).

A before-and-after, site comparison study in 2000–2003 in forest in Queensland, Australia (6) found that installing road underpasses did not reduce numbers of reptiles killed on a highway (although reptile numbers recorded were very low). In the two years after three road underpasses were installed under a highway, two reptiles were counted dead on the road, compared to one reptile in the year before underpass construction. As a comparison, numbers of roadkill reptiles counted on a nearby section of road without an underpass were higher (1 year after installation: 22 reptiles killed; 2 years after: 26 killed). In 2001, a high-altitude highway through rainforest was widened and upgraded to include three concrete wildlife underpasses (3.4 m high, 3.7 m wide). Underpasses incorporated ground cover to simulate the forest floor and arboreal structures (see paper for details). For 12 months prior to underpass construction, two 0.5 km long road transects were surveyed weekly for roadkill by walking either side of the highway. After underpass construction, reptile roadkill was monitored by walking 0.5 km long road transects. Two similar transects were walked on a highway without underpasses 5 km north of the upgraded highway.

A replicated study in 2004–2005 along seven roads in Virginia, USA (7) found that underpasses (including areas under bridges) were used by black rat snakes *Pantherophis obsoletus*. Black rat snakes were observed using at least one of the underpasses (data not provided). In June 2004–May 2005, seven underpasses (including the area under two bridges) were monitored using a camera at each entrance and exit. Photographs were downloaded once a week. Most of the underpasses were designed for water drainage.

A replicated, before-and-after study in 2004–2007 in dry eucalypt woodland in Queensland, Australia (8) found that two road underpasses were used by reptiles from six months after construction was finished. Results were not statistically tested. Six to 12 months after construction of two underpasses, 0.6–1.3 lizards/day and 18–30 months after construction, 0.1–1.0 lizards/day were recorded using the underpasses. The authors reported that most of the reptiles recorded were medium-sized lizards. After construction, one snake *Morelia spilota* was found dead on the road, compared to two snakes (one *Boiga irregularis* and *Cryptophis nigrescens*) beforehand. In 2004, two underpasses were constructed (2.4 m high, 2.5 m wide, 48 m long) containing: a lower level (0.9 m wide), a raised level with rocks (1.6 m wide, 0.4 m above ground), a half log railing, and a wooden shelf (0.25 m wide, 1.2 m above ground) running the length of the underpass. Sand strips (1–2 cm deep, 1 m long, 2.5 m wide) were placed 1–2 m inside the underpasses at both ends and on the shelves (0.5 cm deep, 0.5 m long, 0.25 m wide). Sand was checked for tracks twice weekly in August 2005–February 2006 and monthly in June 2006–June 2007. Animals killed on the road were surveyed by vehicle in the early mornings twice weekly before construction started in April–July 2004 and weekly after construction was completed in February 2005–June 2007. All species larger than a blue-tongued skink *Tiliqua scincoides* were recorded.

A replicated study in 2005–2006 of tunnels in a Wildlife Management Area in New York, USA (9) found that painted *Chrysemis picta* and snapping turtles *Chelydra serpentina* showed some preferences for particular tunnel widths and lengths, but not for different substrates or light levels. Both turtle selected mid-sized diameter tunnels (0.5 m: selected by 39–44% of individuals, 0.6 m: 31–39%) more often than narrower (0.3 m diameter: 6–17%), or wider tunnels (0.8 m: 6–19%). Painted turtles avoided the longest tunnels (3 m long: selected by 10–30% of individuals, 6.1 m: 45%, 9.1 m: 15%), whereas snapping turtles did not show any significant preference for tunnel length (3 m: 20–37%, 6.1 m: 27%, 9.1 m: 17%). Neither turtle showed preferences for substrate type (concrete: 19–37%, gravel: 24–29%, soil: 20–26%, PVC: 20–26%) or light permeability (0% permeability: 26–31%, 0.6%: 14–23%, 1.3%: 15–24%, 5%: 31–36%). Snapping turtles were more likely to not choose any of the tunnels (56%) than painted turtles (16%). Choice arenas had four different PVC culverts radiating out, which painted (74 individuals) and snapping turtles (62) could select to exit through. Separate arenas were constructed to test tunnel diameter (0.3, 0.5, 0.6 or 0.8 m), length (3, 6.1 or 9.1 m), substrate (concrete, soil, gravel or bare PVC) and light (overhead punctures of 0, 0.6, 1.3 or 4% of surface). Turtles were tested individually, once/arena. Trials lasted 15 minutes, after 5 minutes acclimatization, in June–August 2005–2006.

A review in 2010 of studies monitoring 329 road crossing structures in Australia, Europe and North America (10) found that reptiles used crossing structures in 21 of 37 studies. From a total of 37 studies, reptiles used pipes in four of five studies, drainage culverts in nine studies (total number of studies unclear), adapted culverts in four of six studies and bridge underpasses in three of seven studies. Reptiles used a wildlife underpass in one of 13 studies, and in one study they were seen, but did not use the structure. A database (Web of Science) was searched for peer-reviewed, English language studies published in 1998–2008, using a range of keywords relating to roads and wildlife (see paper for details), and reference lists of any papers obtained were also checked.

A replicated study in 2013 along a section of highway through wetlands, rocky outcrops and mixed forest in Ontario, Canada (11) found that few painted turtles *Chrysemys picta* entered a culvert under the highway. In a choice experiment, only 9% (5 of 54) of painted turtles *Chrysemys picta* entered the culvert, whereas 22% (12 of 54) moved away from the culvert and 69% (37 of 54) remained at the entrance. Authors reported that the percentage of turtles entering the culvert was lower than that recorded in a previous arena study away from the highway (47% crossed). In 2013, adult painted turtles (30 females; 24 males) were caught in the wild and brought to a culvert that had been constructed under a highway (2.5 km from capture location). The culvert (and highway) was located between the individuals and their home range. Turtles were allowed to acclimate for 10 minutes in an open box near the entrance to the culvert. The box was then removed and movements were monitored to see if they used the culvert, moved away from it, or did not move.

A replicated, before-and-after study in 2002–2014 on three roads in Ontario, Canada (12) found that two snake species showed similar willingness to enter tunnels. Eastern massasauga rattlesnakes *Sistrurus catenatus* and garter snakes *Thamnophis sirtalis* showed a similar willingness to enter tunnels (rattlesnakes: 18 of 19 entered; garter snakes: 15 of 16) and to complete crossings to the other side (rattlesnakes: 7 of 19; garter snakes: 5 of 16). In 2010–2011, two grate-top tunnels (8.5 m long x 1.2 m wide x 0.5–0.6 m deep) were installed along a road. In 2014, rattlesnakes and garter snakes were caught opportunistically and placed at tunnel entrances to test willingness to enter and complete tunnel crossings.

A site comparison study in 2015 along a grassy verge in Limpopo Province, South Africa (13) found that incidences of reptile road casualties were the same whether or not there were concrete underpasses ('culverts') installed. Results were not statistically tested. When there were culverts, numbers of reptiles killed on the road were the same as when there were not (26 individuals found in both circumstances). A road with 17 culverts was selected. Surveys of reptiles killed on the road were carried out at sunrise in a vehicle along a 12.3 km long stretch of road for 20 consecutive days each in January and February 2015.

A replicated study in 2009–2010 in mixed oak woodland and shrubland in west Andalusia, Spain (14) found that higher numbers and density of underpasses did not reduce numbers of reptile road casualties. Data were reported as statistical model outputs. Spatial distribution of underpasses was not associated with patterns of reptile road casualties, which were found on average 100 m from the nearest underpass. In total 55 reptile carcasses were found (0.1–2.6 reptiles/km). Four roads were surveyed (53 km in total) for reptile carcasses on foot by 1–4 observers during October 2009–January 2010 and April–July 2010 (30 days surveying in total). Each road was surveyed twice. Underpasses were existing culverts or other road drainage tunnels.

A study in 2016 on an inactive railway line in coastal strand habitat in east-central Florida, USA (15) found that gopher tortoises *Gopherus polyphemus* used trenches dug under the rails to move from one side of the railway line to the other. Results were not statistically tested. In total, 68 of 96 tortoise photographs showed animals using the trenches (0.4 tortoise sightings/day). Trenches began to be used within four days of being dug. Two trenches were dug 700 m apart in areas with high tortoise density. A camera trap was set facing the rails (one camera/trench) in May–August 2016 (184 trap days, 92/camera). Individual tortoises were identified using a combination of size, shell patterns, shell shape, and forelimb scale patterns.

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(4) Norman T., Finegan A. & Lean B. (1998) *The role of fauna underpasses in New South Wales.* Proceedings of the 1998 International Conference on Wildlife Ecology and Transportation, Florida Department of Transportation, Tallahassee, Florida USA, 195–208.

(5) Abson R.N. & Lawrence R.E. (2003) *Monitoring the use of the Slaty Creek wildlife underpass, Calder Freeway, Blackforest, Macedon, Victoria, Australia.* Proceedings of the 2003 International Conference on Ecology and Transportation, Center for Transportation and the Environment, North Carolina State University, Raleigh NC, USA, 303–308.

(6) Goosem M., Weston N. & Bushnell S. (2005) *Effectiveness of rope bridge arboreal overpasses and faunal underpasses in providing connectivity for rainforest fauna.* Proceedings of the 2005 International Conference on Ecology and Transportation, Center for Transportation and the Environment, North Carolina State University, Raleigh NC, USA, 304–318.

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(12) Colley M., Lougheed S.C., Otterbein K. & Litzgus J.D. (2017) Mitigation reduces road mortality of a threatened rattlesnake. *Wildlife Research*, 44, 48–59.
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(15) Rautsaw R.M., Martin S.A., Vincent B.A., Lanctot K., Bolt M.R., Seigel R.A. & Parkinson C.L. (2018) Stopped dead in their tracks: the impact of railways on gopher tortoise (*Gopherus polyphemus*) movement and behavior. *Copeia*, 106, 135–143.

5.4. Install overpasses over roads/railways

- **Five studies** evaluated the effects of installing overpasses over roads/railways on reptile populations. Three studies were in Spain[1-3], one was a review of studies in Australia, Europe and North America[4] and one study was in Australia[5].

COMMUNITY RESPONSE (1 STUDY)

- **Community composition (1 study):** One before-and-after, site comparison study in Australia[5] found that the composition of reptile species on a vegetated overpass was more similar to woodland on one side of the overpass than the other.
- **Richness/diversity (1 study):** One before-and-after, site comparison study in Australia[5] found that a vegetated overpass was colonised by two reptile species each year over five years.

POPULATION RESPONSE (1 STUDY)

- **Occupancy/range (1 study):** One before-and-after, site comparison study in Australia[5] found that a vegetated overpass was colonized by 14 of 23 native reptile species and one non-native reptile species.

BEHAVIOUR (4 STUDIES)

- **Use (4 studies):** Three of four studies (including two replicated studies and one review) in Spain[1-3] and Australia, Europe and North America[4] found that overpasses not designed for wildlife were used by lizards and snakes[1,2] and reptiles[4]. The other study[3] found that overpasses not designed for wildlife were not used by snakes or lizards. Two replicated studies in Spain[2,3] found that wildlife overpasses were used by lizards[2] and Ophidians (snakes and legless lizards)[3], and one review in Australia, Europe and North America[4] found that one of 10 wildlife overpasses were used by reptiles. One review of road crossing structures in Australia, Europe and North America[4] found that a rope bridge was not used by reptiles.

Background

To mitigate the effects of roads on wildlife vegetated, vehicle-free overpasses may be constructed. These tend to be constructed for mammals but could also be designed for or used by reptiles.

Wildlife overpasses are constructed to provide safe road and rail crossing opportunities for wildlife. A range of different structures can be used as overpasses including purpose-built "green bridges", on which natural vegetation is established, through to general purpose crossing structures that are accessible to both wildlife and vehicles. Overpasses are often used in combination with wildlife barriers that prevent animals accessing the road and which funnel animals toward the overpasses (see *Install barriers and crossing structures along roads/railways*). Studies summarized within this intervention cover both overpasses created specifically for wildlife and those that were created for other purposes but where information about use of such structures by reptiles is included. Studies mostly report on the use of such structures, such as the number of crossings made, rather than on wider population-level effects of their presence.

See also: *Install tunnels/culverts/underpass under roads/railways*.

A study in 1991–1992 along a high-speed railway within agricultural land in Castilla La Mancha, Spain (1) found that two overpasses not designed for wildlife were used to cross the railway by reptiles. Lizards and snakes were recorded making a total of 112 crossings using two overpasses and 15 underpasses, 7 crossings/100 passage-days on average. Reptile use of overpasses was relatively lower than underpasses (results reported as model outputs, see original paper). Two overpasses (small roads) crossing a 25 km section of a high-speed railway were monitored. The railway was fenced with wire netting on both sides to limit access to the rails. To monitor animal tracks, a layer of sand (3 cm thick and 1 m wide), was put at one entrance to each overpass, and tracks were monitored for 15–22 days/month between September 1991 and July 1992.

A replicated study in 2002 of a highway in Zamora, Spain (2, same experimental set-up as 3) found that wildlife and other overpasses were used by reptiles. Lacertid lizards (*Lacerta* spp. and *Podarcis* spp.) were recorded crossing wildlife overpasses (0.5 crossings/day/structure) and lacertids and ophidians (snakes and legless lacertids) were also recorded crossing other overpasses, such as rural tracks (lacertids: 0.4 crossings/day/structure, ophidians: 0.1). Two wildlife overpasses (16 m wide, 60 m long) and 16 general overpasses (rural tracks, 7–8 m wide, 58–62 m long) were monitored along a 72 km section of the A-52 motorway. The motorway had barrier fencing along its length. Marble dust (1 m wide across) was used to record animal tracks for 10 days in June–September 2002. Camera traps were installed on some overpasses.

A replicated study in 2001 of a highway in Zamora province, Spain (3 same experimental set-up as 2) found that wildlife, but not other overpasses were used by some reptiles. Ophidians (snakes and legless lacertids) were recorded crossing wildlife overpasses (0.3 crossings/day/structure) but not other overpasses, such as rural tracks. Lacertid lizards (*Lacerta* spp. and *Podarcis* spp.) were not recorded using any overpasses. Four wildlife overpasses (15–20 m wide, 60–62 m long) and six general overpasses (rural tracks, 7–8 m wide, 58–65 m long) were monitored along the A-52 motorway. The motorway had barrier fencing along its length. Marble dust (1 m wide cross) was used to record animal tracks daily for 10 days in March–June 2001.

A review in 2010 of studies monitoring 329 road crossing structures in Australia, Europe and North America (4) found that reptiles used overpass crossing structures in three of 15 studies. Reptiles were recorded using overpasses in two of 15 studies and wildlife overpasses in one of 10 of the studies (in one study reptiles were present but did not use the structure). One study of a rope bridge did not record any reptiles. The use of overpasses, wildlife overpasses and canopy-rope bridges by wildlife was reported for 15 studies.

A before-and-after, site comparison study in 2005–2010 in eucalypt forest and woodland next to a highway in Queensland, Australia (5) found that a vegetated overpass was colonised by reptile species native to the area. Fourteen of 23 native reptile species found in the area were captured on the vegetated overpass. One non-native reptile species was captured on the overpass but not in adjacent woodland. Capture data over time indicated that the overpass had been colonized at a rate of two species/year. Community composition on the overpass tended to be more similar to woodland on one side of the overpass than the other. A vegetated, fenced overpass was constructed in 2005 and planted with native vegetation sourced from local woodlands. Six woodland sites <1 km from and on both sides of the vegetated overpass were surveyed from June 2005–February 2010 and one site on the overpass was surveyed from February 2006–February 2010. Reptile data were collected from pitfall traps constructed of 15 m drift fences and three 20 L buckets, and hand searches for three days and two nights every two months. Animals were not marked and released immediately after identification.

(1) Rodriguez A., Crema G., & Delibes M. (1996) Use of non-wildlife passages across a high speed railway by terrestrial vertebrates. *Journal of Applied Ecology*, 33, 1527–1540.

(2) Mata C., Hervás I., Herranz J., Suarez F. & Malo J.E. (2005) Complementary use by vertebrates of crossing structures along a fenced Spanish motorway. *Biological Conservation*, 124, 397–405.

(3) Mata C., Hervás I., Herranz J., Suárez F. & Malo J.E. (2008) Are motorway wildlife passages worth building? Vertebrate use of road-crossing structures on a Spanish motorway. *Journal of Environmental Management*, *88,* 407–415.

(4) Taylor B.D. & Goldingay R.L. (2010) Roads and wildlife: impacts, mitigation and implications for wildlife management in Australia. *Wildlife Research*, 37, 320–331.

(5) McGregor M.E., Wilson S.K. & Jones D.N. (2015) Vegetated fauna overpass enhances habitat connectivity for forest dwelling herpetofauna. *Global Ecology and Conservation*, 4, 221–231.

5.5. Manually remove reptiles from roads

- **One study** evaluated the effect on reptile populations of manually removing reptiles from roads. This study was in the USA[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Survival (1 study):** One study in the USA[1] reported that when turtles were being removed from a road following installation of a fence and artificial nesting mounds, fewer turtles were killed on the road than in the year before any interventions began.

BEHAVIOUR (0 STUDIES)

Background

Many reptiles are killed by vehicles, particularly when crossing roads when moving to and from breeding habitats. In some areas, local volunteers may try to reduce deaths by collecting reptiles and releasing them on the other side of the road.

Ideally evidence of the effectiveness of this intervention would consist of survival rates, counts of animals in the population or numbers killed on the road before and after or at sites with and without human assistance. However, such evidence is rarely available.

For other interventions that involve engaging volunteers to help manage reptiles or their habitats see *Education and awareness raising – Engage local communities in conservation activities*.

A study in 1999–2008 on a roadside verge along a river bank in Pennsylvania, USA (1) found that when turtles were actively moved off a road after a chain-link fence was installed along a highway and artificial nest mounds were created on the non-road side of the fence, fewer female northern map turtles *Graptemys geographica* were killed on the road compared to before any interventions began. Results were not statistically tested. In the 2nd to 8th year after a fence was installed on a new major highway, when turtles were being actively removed from the road, 0–5 map turtles died on the road/year (no data for other species). In the first year after a fence was installed, 10 northern map turtles were killed on the road, and in the years before installation 50 turtles were killed on the road (total included a small number of wood turtles *Glyptemus insculpta* and snapping turtles *Chelydra serpentina*). The authors reported that most deaths were gravid female turtles. In 2000, a fence (1 m high and 1,150 m long) was installed on the river side of the highway to prevent turtles from crossing the road to access nesting habitat. Mounds of sand aimed at providing alternative nesting habitat were added to the river side of the fence in 2000–2001. After the first year, the fence was extended by 300 m to prevent turtles from going around it and crushed shale was added to the sand mounds and turtles were actively moved off the road. Turtle deaths on the road were counted from 1999 (the first year after a new highway opened) to 2008 (excluding 2004).

(1) Nagle R.D. & Congdon J.D. (2016) Reproductive ecology of *Graptemys geographica* of the Juniata river in Central Pennsylvania, with recommendations for conservation. *Herpetological Conservation and Biology*, 11, 232–243.

5.6. Use signage to warn motorists about wildlife presence

- **Five studies** evaluated the effects of using signage to warn motorists of wildlife presence on reptile populations. Three studies were in the USA[1,2,5] and one was in each of Dominica[3] and Canada[4].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (5 STUDIES)

- **Survival (5 studies):** One of two before-and-after studies (one replicated and controlled) in the USA[1,2] found that installing road signs reduced road mortalities of massasaugas in autumn but not summer. The other study[2] found that installing road signs did not reduce road mortalities of painted or Blanding's turtles. Two before-and-after studies (one replicated) in Canada[4] and the USA[5] found that a combination of installing road signs with either fencing and tunnels[4] or a hybrid nestbox-fencing barrier[5] resulted in fewer road mortalities of massasaugas[4] and diamondback terrapins[5]. One before-and-after study in Dominica[3] found that a combination of using road signs and running an awareness campaign resulted in fewer road mortalities of Antillean iguanas.

BEHAVIOUR (0 STUDIES)

Background

Wildlife crossing signs alert drivers to the potential presence of wildlife on or near a road. They are designed to encourage drivers to be more alert and/or reduce the speed of their vehicle, with the goal of reducing animal-vehicle collisions. Motorists may become habituated to signs if they are present all year round, are too common or look similar to other signs. Solutions may be to use temporary seasonal signs, animated signs, flashing lights or flags to catch the attention of drivers.

A before-and-after study in 1981–1982 on a road through a prairie in Missouri, USA (1) reported that when road signs were installed to warn motorists of snakes, a lower percentage of total massasaugas *Sistrurus catenatus* found were dead on the road in one of two seasons compared to when signs were not present. Results were not statistically tested. Of the total number of snakes found during the study (172 individuals), the percentage that were dead on the road was similar before (19% dead) and after signs were installed (24%) in summer, but lower after signs were installed in autumn (after 13%; before 32%). Road signs warning motorists of snakes were installed in 1981 (month not given). Surveys for snakes were conducted on a prairie and bordering roads and dykes, and trapping was carried out using drift fences with wire-mesh funnel traps (number and timing of surveys not provided).

A replicated, controlled, before-and-after study in 2008–2010 on 18 roads through swamps and wetlands in New York, USA (2) found that road signs did not decrease painted Chrysemys picta and Blanding's Emydoidea blandingii turtle road mortality. The percentage of turtles encountered that were dead was similar on roads with signs (2009: 49 of 72, 68%; 2010: 20 of 31, 65%) and roads without signs (2008: 40 of 71, 56%; 2009: 28 of 53, 53%; 2010: 16 of 28, 57%). Road signs warning drivers of turtles were installed during the nesting season (1 June–1 July) on five roads in 2009 and nine roads in 2010. Driving (daily) and walking (100 m transect, 1–2 times/week) surveys were conducted to count the number of turtles encountered (dead and alive) before signs were installed in 2008 (9 roads) and after signs were installed in 2009 (Signs: 5 roads; no signs: 5 roads) and 2010 (Signs: 9 roads; no signs: 9 roads). Dead animals were removed to prevent double counting. Results from 2010 include only driving surveys.

A before-and-after study in 2008–2010 on coastal roads on the Caribbean Sea side of Dominica (3) found that using road signs and running an awareness campaign reduced lesser Antillean iguana *Iguana delicatissima* road mortality by half. After putting up road signs and running an awareness campaign, lesser Antillean iguana road mortality reduced by approximately 50% (0.3 fatal collisions/day) on coastal roads compared to beforehand (0.6 fatal collisions/day). An awareness campaign about protecting iguanas was carried out in May 2008–June 2010. On 1 July 2009, road signs asking people to slow for iguanas were put up on coastal roads near known nesting locations (see original paper for details). The campaign included lectures at schools, presentations to government employees, radio and television interviews and distributing bumper stickers across the island asking people to slow down for iguanas. Two coastal road segments (11–29 km long) were surveyed for iguanas during the nesting season from April 2008–June 2010, 122 days before signs were put up and 94 days afterwards.

A replicated, before-and-after study in 2002–2014 on three roads in Ontario, Canada (4) found that eastern massasauga rattlesnake *Sistrurus catenatus* road mortality was reduced after signs for motorists, guide fencing and tunnels were installed. The number of rattlesnakes found, both dead or alive on roads decreased after sign, fence and tunnel installation (dead: 6, alive: 14) compared to before installation (dead: 41, alive: 68) and during installation (dead: 15, alive: 37). In 2007–2013, four signs to encourage motorists to slow down for snakes were installed near known snake road crossing locations (precise installation dates not provided), and three sets of mesh barrier fences were installed (600 m–1 km apart; 600–900 m of fencing on one or both sides of the road). In 2010–2011, within each of the two sets of fencing on both sides of the road, two grate-top tunnels (8.5 m long x 1.2 m wide x 0.5–0.6 m deep) were installed. Eastern massasauga rattlesnakes on the road were surveyed from May to October by car before (2005–2007) and during installation (2008–2012), and by bicycle after installation (2013–2014).

A before-and-after study in 2009–2015 on saltmarsh in Georgia, USA (5) found that adding a flashing terrapin-warning sign to alert motorists and partially fencing a causeway reduced the likelihood of mortality for diamondback terrapins *Malaclemys terrapin* crossing the road. When the flashing signs and hybrid nestbox-fence barrier were installed on a road, survival of crossing female diamondback terrapins increased from 24% to 53% (data reported as statistical model outputs). In 2011, a 22 m hybrid nestbox-fence barrier was built along a 9 km long causeway. In 2013, two terrapin crossing signs with flashing warning beacons were added to warn motorists entering a 6 km stretch of causeway. The signs were activated for 2 hours/day during peak terrapin crossing times. Terrapins were surveyed on the causeway and in adjacent creeks during the nesting season (May–July) in 2009–2015.

(1) Seigel R.A. (1986) Ecology and conservation of an endangered rattlesnake, *Sistrurus catenatus*, in Missouri, USA. *Biological Conservation*, 35, 333–346.
(2) Johnson G. (2012) *Testing the effectiveness of turtle crossing signs as a conservation measure.* Final Report prepared for St. Lawrence River Research and Educational Fund, New York Power Authority, New York, USA.
(3) Knapp C.R., Prince L. & James A. (2016) Movements and nesting of the Lesser Antillean Iguana (*Iguana delicatissima*) from Dominica, West Indies: Implications for conservation. *Herpetological Conservation and Biology*, 11, 154–167.
(4) Colley M., Lougheed S.C., Otterbein K., Litzgus J.D. (2017) Mitigation reduces road mortality of a threatened rattlesnake. *Wildlife Research*, 44, 48–59.
(5) Crawford B.A., Moore C.T., Norton T.M. & Maerz J.C. (2018) Integrated analysis for population estimation, management impact evaluation, and decision-making for a declining species. *Biological Conservation*, 222, 33–43.

5.7. Reduce legal speed limit

- We found no studies that evaluated the effects of reducing the legal speed limit on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

High vehicle speed is generally considered to be a substantial contributing factor in wildlife-vehicle collisions. Slower road speeds allow drivers to identify and avoid wildlife, as well as allowing reptiles the time necessary to cross roads. Speed limits can be reduced in areas where there are high numbers of collisions, either permanently or during seasonal migrations.

See also *Use signage to warn motorists about wildlife presence.*

5.8. Limit or exclude off-road vehicle use

- **Two studies** evaluated the effects of limiting or excluding off-road vehicle use on reptile populations. Both studies were in the USA[1,2].

COMMUNITY RESPONSE (1 STUDIES)

- **Richness/diversity (1 studies):** One replicated, site comparison study[1] found that restricting access of off-road vehicles and sheep had mixed effects on lizard species richness.

POPULATION RESPONSE (2 STUDIES)

- **Abundance (2 studies):** One of two replicated, site comparison studies (including one randomized study) in the USA[1,2] found that areas where off-road vehicles were completely excluded using fencing that also excluded livestock grazing had higher densities of Agassiz's desert tortoises compared to areas with some restrictions or no restrictions[2]. The other study[1] found that restricting off-road vehicle and sheep access had mixed effects on lizard abundance.
- **Survival (1 study):** One replicated, randomized, site comparison study in the USA[2] found that in areas where off-road vehicles were completely excluded, death rates of

Agassiz's desert tortoises were lower than in areas with some restrictions or no restrictions.

BEHAVIOUR (0 STUDIES)

Background

Continued and uncontrolled use off-road vehicles may damage vegetation and soil characteristics. The impact on wildlife is less studied, although impacts on birds, mammals and reptiles are commonly reported (Vollmer *et al*. 1977).

Vollmer A.T., Maza B.G., Medica P.A., Turner F.B. & Bamberg S.A. (1977) The impact of off-road vehicles on a desert ecosystem. *Environmental Management*, 1, 115–129.

A replicated, site comparison in 1994–1996 in desert shrub and grassland in south-central California, USA (1), found that lizard abundance and species richness was higher inside a fenced protected area that excluded vehicles and sheep, compared to outside of the fence, depending on survey month and plot. Lizard abundance was higher in three of six survey comparisons in a fenced protected area with restricted vehicle use (4–10 lizards/transect) compared to outside of it (2–4 lizards/transect) but similar in the remaining three comparisons (inside protected area: 2–5 lizards/transect; outside protected area 1–3 lizards/transect; see original paper for details). Lizard species richness was higher in one of six comparisons inside the protected area (2 species/transect) compared to outside of it (1 species/transect) but similar in the remaining five comparisons (inside protected area: 2–3 species/transect; outside protected area: 1–3 species/transect; see original paper for details). In 1994, two sites were selected near the north-eastern and southern boundary of the Desert Tortoise Research Natural Area (where off-road vehicles were prohibited from 1973, sheep grazing prohibited from 1978 and the boundary was fenced in 1980). Two 2.25 ha plots were established/site: one ≥400m inside the boundary and one outside the boundary (used by off-road vehicles until 1980 and grazed by sheep until 1994). In each plot, lizards were surveyed using 1.25 km transects in July 1994 and May and July 1995 (six surveys/site).

A replicated, randomized, site comparison study in 2011 in desert shrub and grassland in California, USA (2) found that Agassiz's desert tortoises Gopherus agassizii were more abundant and had a lower mortality rate in a protected area fenced to exclude recreational vehicle use and livestock grazing. Desert tortoise densities were approximately six-times higher in the most protected area, the Tortoise Natural Area (15 tortoises/km^2) than designated tortoise critical habitat with some vehicle restrictions (2 tortoises/km^2) and four-times higher than on private lands with no vehicle restrictions (4 tortoises/km^2). Tortoise annual death rates over the preceding four years were estimated as lowest in the Tortoise Natural Area (3% mortality/year) compared to private lands (6%) or in tortoise critical habitat (20%, results were not statistically tested). Tortoises were surveyed in 240 1 ha plots across three different management areas (80 plots/area): Tortoise Natural Area (1973: closed to recreational vehicles; 1980: fully enclosed and closed to mining and livestock grazing, 2010: 12 km of fencing extended to prevent tortoises leaving), tortoise critical habitat areas (1994: recreational vehicle use restricted but not enforced with some annual closures, 1990: closed to sheep grazing) and private lands (unregulated sheep grazing, intensive recreational vehicle use, hunting and trash dumping). In April–May 2011 plots were surveyed on foot twice in a day for live or dead tortoises and field signs.

(1) Brooks M. (1999) Effects of protective fencing on birds, lizards, and black-tailed hares in the western Mojave Desert. *Environmental Management*, 23, 387–400.
(2) Berry K.H., Lyren L.M., Yee J.L. & Bailey T.Y. (2014) Protection benefits desert tortoise (*Gopherus agassizii*) abundance: the influence of three management strategies on a threatened species. *Herpetological Monographs*, 28, 66–92.

5.9. Use road closures

- **One study** evaluated the effects of using road closures on reptile populations. This study was in Canada[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One replicated study in Canada[1] found that closed roads were not used more by Blanding's turtles than unclosed roads.

Background
Reptiles are vulnerable to road mortality, particularly during the breeding season when activity levels increase and roads may bisect nesting habitat. In some areas, roads may be closed to protect hotspots for reptile movements.

A replicated study in 2010 in wetlands and forests bisected by roads along the Ottawa River in southern Québec, Canada (1) found that closed roads were not used more by Blanding's turtles *Emydoidea blandingii* compared to roads open to vehicle traffic. Blanding's turtles showed similar levels of use of roads closed (0.9 crossings/individual) and open to vehicle traffic (1.1 crossings/individual). Twenty-four of 52 turtles crossed roads. Fifty-two Blanding's turtles (22 females, 24 males, 6 juveniles) were captured by hand or using hoop nets and a radio transmitter was attached to their shell. All turtles were tracked every 2–4 days from May to August 2010.

(1) Proulx C.L., Fortin G. & Blouin-Demers G. (2014) Blanding's turtles (*Emydoidea blandingii*) avoid crossing unpaved and paved roads. *Journal of Herpetology*, 48, 267–271.

5.10. Alter road surfaces

- **One study** evaluated the effects of altering road surfaces on reptile populations. This study was in Canada[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One replicated study in Canada[1] found that paved roads were not used more by Blanding's turtles than unpaved roads.

Background
Due to the way that reptiles regulate their body temperature using external sources, it is possible that sun-warmed roads surfaces may contribute to road-related mortality. Gravel roads are cooler than surrounding natural habitats and may offer an intervention option to prevent reptiles from crossing roads (Shine *et al*. 2004).

Shine R., Lemaster M., Wall M., Langkilde T. & Mason R. (2004) Why did the snake cross the road? Effects of roads on movement and location of mates by garter snakes. *Ecology and Society,* 9, 9.

A replicated study in 2010 in wetlands and forests bisected by roads along the Ottawa River in southern Québec, Canada (1) found that Blanding's turtles *Emydoidea blandingii* did not cross paved roads more compared to unpaved roads. Blanding's turtles showed similar levels of use of paved roads (0.1 crossings/individual) compared to unpaved roads (1.0 crossings/individual). Twenty-four of 52 turtles crossed roads. Fifty-two Blanding's turtles (22 females, 24 males, 6 juveniles) were captured by hand or using hoop nets and a radio transmitter was attached to their shell. All turtles were tracked every 2–4 days from May to August 2010.

(1) Proulx C.L., Fortin G. & Blouin-Demers G. (2014) Blanding's turtles (*Emydoidea blandingii*) avoid crossing unpaved and paved roads. *Journal of Herpetology*, 48, 267–271.

5.11. Retain/maintain road verges as habitat

- We found no studies that evaluated the effects of retaining/maintaining road verges as habitat on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background
Roads can damage or destroy grassland habitats that host reptiles, but roadside verges may be used by reptiles (e.g. Carthew *et al*. 2013). Maintaining road verges with reptiles in mind may contribute towards mitigating habitat loss.

Carthew S.M., Garrett L.A. & Ruykys L. (2013) Roadside vegetation can provide valuable habitat for small, terrestrial fauna in South Australia. *Biodiversity and conservation*, 22, 737–754.

5.12. Limit road construction in important habitats

- We found no studies that evaluated the effects of limiting road construction in important habitats on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background
Limiting road construction is a legislative intervention requiring that impacted species or habitats be considered at the planning stages of road or service corridors. It may include preventing road construction altogether, or changing the road design to avoid key habitats. For example, a case study in North Carolina, USA

suggested that a road was moved 100 m to avoid a timber rattlesnake *Crotalus horridus* rookery and a high density of eastern box turtles *Terrapene carolina* (Andrews *et al*. 2006).

Andrews K.M., Gibbons J.W. & Jochimsen D.M. (2006) *Literature synthesis of the effects of roads and vehicles on amphibians and reptiles*. Federal Highway Administration (FHWA), US Department of Transportation, Report No. FHWA-HEP-08-005. Washington, D.C.

Utility & Service Lines

5.13. Install crossings over/under pipelines

- We found no studies that evaluated the effects of installing crossings over/under pipelines on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Pipelines can extend hundreds of kms and may represent substantial barriers to reptile movements if they lie at or just above the surface of the ground. Crossing points can be either elevated sections of pipe with space for reptiles to pass beneath, buried sections or sections with crossing ramps constructed over the pipe.

Aquatic Transport Corridors & Boats

5.14. Limit vessel numbers

- We found no studies that evaluated the effects of limiting vessel numbers on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Collisions with boats or their motors can kill or severely injure reptiles. Increases in watercraft activity in areas developed for human recreation have been shown to greatly increase the likelihood of major shell injuries and limb amputations in some species (e.g. Cecala *et al.* 2009). Limiting numbers of vessels may reduce collision rates.

Cecala K.K., Gibbons J.W. & Dorcas M.E. (2009) Ecological effects of major injuries in diamondback terrapins: implications for conservation and management. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 19, 421–427.

5.15. Limit vessel speeds

- **Three studies** evaluated the effects of limiting vessel speeds on reptiles. One study was in each of Australia[1], Costa Rica[2] and the USA[3].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Survival (2 studies):** One replicated, site comparison study in Costa Rica[2] found that in waterways with enforced speed limits, fewer spectacled caiman were found dead with boat-related injuries compared to waterways with no speed limits. One replicated study in the USA[3] found that vessels travelling at lower speeds caused fewer catastrophic injuries to artificial loggerhead turtle shells, though vessels with jet motors caused no catastrophic injuries at any speed tested.
- **Condition (1 study):** One replicated, site comparison study in Costa Rica[2] found that in waterways with enforced speed limits, fewer spectacled caiman were found with boat-related injuries compared to waterways with no speed limits.

BEHAVIOUR (1 STUDY)

- **Behaviour change (1 study):** One replicated study in Australia[1] found that green turtles were more likely to flee from vessels travelling at lower speeds.

Background

Turtle collisions with boats are of major concern due to the high rate of strike injury found in all species, both freshwater (Bennett & Litzgus 2014) and oceanic (Hazel *et al*. 2007, Denkinger *et al.* 2013). Establishing speed limits for watercraft in important areas has been found to benefit some aquatic animals by giving them more time to avoid collisions with approaching vessels (Nowacek *et al.* 2004, Laist & Shaw 2006) and may similarly benefit aquatic reptiles.

Bennett A.M. & Litzgus J.D. (2014) Injury rates of freshwater turtles on a recreational waterway in Ontario, Canada. *Journal of Herpetology*, 48, 262–266.

Denkinger J., Parra M., Muñoz J.P., Carrasco C., Murillo J.C., Espinosa E., Rubianes F. & Koch V. (2013) Are boat strikes a threat to sea turtles in the Galapagos Marine Reserve? *Ocean & coastal management*, 80, 29–35.

Hazel J., Lawler I.R., Marsh H. & Robson S. (2007) Vessel speed increases collision risk for the green turtle *Chelonia mydas*. *Endangered Species Research,* 3, 105–113.

Laist D.W. & Shaw C. (2006) Preliminary evidence that boat speed restrictions reduce deaths of Florida Manatees. *Marine Mammal Science*, 22, 472–479.

Nowacek S.M., Wells R.S., Owen E.C., Speakman T.R., Flamm R.O. & Nowacek D.P. (2004) Florida Manatees, *Trichechus manatus latirostris*, respond to approaching vessels. *Biological Conservation* 119, 517–523.

A replicated study in 2004 in shallow oceanic water off the coast of Queensland, Australia (1) found that green turtles *Chelonia mydas* were more likely to flee a vessel driven at slower speeds. Of 1,819 turtle encounters when the turtle was on the sea floor, 60% were able to flee a slow-moving boat (416 of 694 turtles fled), 22% at a moderate speed (136 of 620 turtles fled) and 4% at a fast speed (20 of 505 turtles fled). This trend was statistically significant only when turtles were within 6 m offset of the vessel (see original paper). Turtles in the water column or on the surface also tended to show a reduced flight response at faster vessel speeds (small sample size precluded statistical analysis, see original paper). A 6 m aluminium boat with a 40 hp outboard motor was driven at three speeds (4, 11 and 19 km/h) on a transect 5 km parallel to the shoreline about 200–450 m from the shore in <5 m of water in the 3 hours before and after noon. The behaviour of turtles sighted within 10 m of the boat were recorded by a spotter.

A replicated, site comparison study in 2006 in a canal system in Limon Province, Costa Rica (2) found that waterways with enforced speed limits had lower numbers of injured spectacled caiman *Caiman crocodilus fuscus*. No injured spectacled caiman were caught in waterways with enforced speed limits (injured: 0 individuals; non-injured: 24 individuals), whereas 37% of spectacled caiman caught in waterways without enforced speed limits had boat-related injuries (injured: 11 individuals; non-injured: 19 individuals), of which two died. Caiman were surveyed in April–June 2006 in three waterways with enforced speed limits (idle–slow) and three without enforced speed limits (high speeds up to 40 km/hour). Adult caiman (1.0–2.5 m long) were caught at night and checked for scars or injuries. Mortalities from boat propellers were recorded.

A replicated study in 2009 in an abandoned sand quarry in Georgia, USA (3) found that lower vessel speeds reduced catastrophic injuries to artificial loggerhead turtle *Caretta caretta* shells. Catastrophic injuries to artificial turtle shells occurred less frequently at lower speeds than higher speeds on unmodified vessels (7 km/hr: four of 10 turtles; 40 km/hr: 10 of 10 turtles) and vessels with propeller guards (7 km/hr: one of 10; 40 km/hr: nine of nine). With a jet outbound motor or an inboard jet motor (on a jet ski) none of 40 turtles were damaged at 7 or 40 km/hr. Injury rates were similar regardless of the position of the artificial shell in the water. A 90 hp, 4-stroke outboard motor with a three-bladed stainless-steel propeller was mounted on a 5.8 m skiff and driven at 7 or 40 km/h over surface level or propeller-depth (48 cm) fibreglass model loggerhead turtle shells (5 trials/speed/turtle depth). One of two propeller guard designs were then added to the same skiff: a horizontal-fin (Hydroshield®), or a stainless-steel cage (Prop Buddy®) and driven over propeller-depth artificial turtle shells (7 km/h: 5 trials/guard; 40 km/h: 4–5 trials/guard). A personal watercraft (jet ski) with an inboard jet motor and the 5.8 m skiff modified with an 80 hp, jet-drive outboard motor were also both driven at 7 and 40 km/h over surface-level or propeller-depth artificial loggerhead turtle shells (5 trials/speed/vessel/turtle depth). Injuries to artificial shells were classified as catastrophic if they would have killed a real sea turtle.

(1) Hazel J., Lawler I.R., Marsh H. & Robson S. (2007) Vessel speed increases collision risk for the green turtle *Chelonia mydas*. *Endangered Species Research*, 3, 105–113.

(2) Grant P.B.C. & Lewis T.R. (2010) High speed boat traffic: A risk to crocodilian populations. *Herpetological Conservation and Biology*, 5, 456–460.

(3) Work P.A., Sapp A.L., Scott D.W. & Dodd M.G. (2010) Influence of small vessel operation and propulsion system on loggerhead sea turtle injuries. *Journal of Experimental Marine Biology and Ecology*, 393, 168–175.

5.16. Establish protocols to reduce collisions

- We found no studies that evaluated the effects of establishing protocols to reduce collisions on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Protocols involving visual observation networks, dedicated onboard observers, and predictive modelling may be employed to reduce vessel collisions with sea turtles by alerting skippers to the presence of turtles and allowing alternative navigation routes to be used.

5.17. Train vessel operators on appropriate avoidance techniques to reduce collisions

- We found no studies that evaluated the effects on reptile populations of training vessel operators on appropriate avoidance techniques to reduce collisions.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Sea turtle collisions with vessels may be avoided if animals are detected and avoidance measures are carried out by the vessel operator. Training vessel crew to detect and identify sea turtles and using dedicated observers on vessels may help avoid collisions in daylight on manoeuvrable vessels (Schoeman *et al*. 2020).

Schoeman R.P., Patterson-Abrolat C. & Plön S. (2020) A global review of vessel collisions with marine animals. *Frontiers in Marine Science*, 7, 292.

5.18. Use technology and reporting systems to avoid collisions

- We found no studies that evaluated the effects on reptile populations of using technology and reporting systems to avoid collisions.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Most technological solutions for avoiding collisions have been deployed to protect marine mammals but may present viable solutions for protecting sea turtles as well (Schoeman *et al*. 2020). Systems are typically used to let vessel operators know about the presence of high densities or recent sightings of animals in real time to allow operators to reduce speeds or change course.

Schoeman R.P., Patterson-Abrolat C. & Plön S. (2020) A global review of vessel collisions with marine animals. *Frontiers in Marine Science*, 7, 292.

5.19. Use visual or acoustic deterrents to discourage reptiles from approaching vessels

- We found no studies that evaluated the effects on reptile populations of using visual or acoustic deterrents to discourage reptiles from approaching vessels.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Acoustic and visual deterrents have been developed to cause sea turtles to flee from vessels (e.g. Lenhardt 2002) but little is known about their efficacy in practice. However, practitioners need to consider the potential for acoustic trauma resulting from sound transmissions, the habituation of target species to sounds and the potential to disturb and displace other non-target marine fauna (Schoeman *et al*. 2020).

For studies describing the effects of using visual deterrents or adding lights to fishing gear, see *Threat: Biological resource use – Use visual deterrents on fishing gear* and *Add lights to fishing gear*.

Lenhardt M.L. (2002) *Marine Turtle Acoustic Repellent/Alerting Apparatus and Method*. US Patent No 6,388,949 B1. Arlington, VA: Sound Technique Systems.

Schoeman R.P., Patterson-Abrolat C., & Plön S. (2020) A global review of vessel collisions with marine animals. *Frontiers in Marine Science*, 7, 292.

5.20. Modify vessels to reduce or prevent injuries to reptiles from collisions

- **Two studies** evaluated the effects on reptile populations of modifying vessels to reduce or prevent injuries to reptiles from collisions. Both studies were in the USA[1a,1b].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Survival (2 studies):** One controlled study[1a] found that using a horizontal-fin propeller guard or a cage propeller guard did not reduce catastrophic injuries to artificial loggerhead turtle shells compared to using no guard, but that the types of injuries sustained were different. One controlled study[1b] found that using a jet drive outboard motor reduced catastrophic injuries to artificial loggerhead turtle shells compared to using a standard outboard motor.

BEHAVIOUR (0 STUDIES)

Background

Many sorts of propeller guards or design changes have been suggested to prevent human injuries, ranging from polypropylene to stainless-steel cages or baffles on outboard motors and these may also help prevent or limit injuries to aquatic reptiles.

A controlled study in 2009 in a sand quarry in Georgia, USA (1a) found that using propeller guards did not reduce catastrophic injuries to artificial loggerhead turtle *Caretta caretta* shells compared to an unmodified propeller. Results were not statistically tested. At 7 km/hr, a cage propeller guard caused none of five artificial loggerhead turtle shells catastrophic damage, whereas a horizontal-fin propeller guard or no guard caused one of five shells to be catastrophically damaged. At 40 km/hr, vessels with both types of guard, or no propeller guard caused catastrophic injuries to all shells in all trials (horizontal-fin: five of five shells damaged; cage guard: four of four; no guard: five of five). The authors reported that the types of injuries sustained were different when guards were used (see paper for details). A 90 hp, 4-stroke outboard motor with a three-bladed stainless steel propeller was mounted on a 5.8 m flat-bottomed skiff and driven at 7 and 40 km/h over propeller-depth (48 cm) fibre-glass model loggerhead turtle shells using either one of two propeller guards: a horizontal-fin mounted below the propeller (Hydroshield®; 5 trials each at 7 and 40 km/h), or a stainless-steel cage surrounding the propeller (Prop Buddy® 5 trials at 7 km/h and 4 trials at 40 km/h), or no guard at all (5 trials/speed). Injuries to the artificial shell were classified as catastrophic if they would have killed a real sea turtle.

A controlled study in 2009 in a sand quarry in Georgia, USA (1b) found that using a jet drive outboard motor on a skiff reduced catastrophic injuries to artificial loggerhead turtle *Caretta caretta* shells compared to a standard outboard motor. Artificial loggerhead turtle shells hit by a skiff with a jet drive outboard motor received fewer catastrophic injuries regardless of speed or turtle depth in the water (0 catastrophic injuries in 20 trials) compared to shells hit by a skiff with a standard outboard motor (14 catastrophic injuries in 20 trials). A 5.8 m flat-bottomed vessel was fitted with either a 90 hp, 4-stroke outboard motor with a three-bladed stainless-steel propeller or an 80 hp, jet drive outboard motor and driven at 7 and 40 km/h over fibre-glass model loggerhead turtle shells (see original paper for details). Shells were placed on the surface or floating 48 cm below the surface. Five trials were carried out for each motor type at each speed and each turtle depth (40 total trials). Injuries to the artificial shell were classified as catastrophic if they would have killed a real sea turtle.

(1) Work P.A., Sapp A.L., Scott D.W. & Dodd M.G. (2010) Influence of small vessel operation and propulsion system on loggerhead sea turtle injuries. *Journal of Experimental Marine Biology and Ecology*, 393, 168–175.

6. Biological resource use

Background

Biological resource use includes the deliberate consumptive use of reptiles through hunting, trapping and collection of eggs; the unintentional harvesting of reptiles in fisheries that are targeting other species (sometimes referred to as 'bycatch'); and the destruction to reptile habitats that can be caused by both deliberate and unintentional collection or harvesting of reptiles (e.g. Goode *et al.* 2005). Logging and wood harvesting are also included, which pose an indirect threat to reptiles through habitat destruction and fragmentation and disturbance.

Goode M.J., Horrace W.C., Sredl M.J. & Howland J.M. (2005) Habitat destruction by collectors associated with decreased abundance of rock-dwelling lizards. *Biological Conservation*, 125, 47–54.

Hunting and collecting animals

6.1. Regulate wildlife harvesting

- **Four studies** evaluated the effects of regulating wildlife harvesting on reptile populations. One study was in each of Costa Rica[1], Australia[2], Indonesia[3] and Japan[4].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Abundance (2 studies):** One before-and-after study in Australia[2] found that following legal protection and harvest regulations, the density of saltwater crocodile populations increased. One before-and-after study in Japan[4] found that following regulation of the green turtle harvest in combination with allowing harvested turtles to lay eggs prior to being killed, the number of nesting females tended to be higher.
- **Reproductive success (1 study):** One before-and-after study in Japan[4] found that following regulation of the green turtle harvest in combination with allowing harvested turtles to lay eggs prior to being killed, the number of hatchlings produced in natural nests tended to be higher.
- **Condition (1 study):** One before-and-after study in Australia[2] found that following legal protection and harvest regulations, the average size of crocodiles increased.

BEHAVIOUR (0 STUDIES)

OTHER (2 STUDIES)

- **Human behaviour change (2 studies):** One replicated study in Costa Rica[1] found that in an area with a legalized turtle egg harvest run by a community cooperative, a majority of people reported a willingness to do more to protect sea turtles. One study in Indonesia[3] reported that quotas to regulate wildlife harvesting did not limit the number of individuals of three reptile species that were harvested and exported.

Background

The harvesting of reptiles and their eggs may be regulated by preventing trade altogether, or by setting harvesting and export quotas that are designed to enable the population to reach or remain at a particular level. However, it should be noted that in some cases the basis for determining such quotas is not clear, and numbers may not be evidence based (Auliya 2010). Whilst many hunting systems use quotas, the studies included here are those based on species with particular conservation concerns rather than where quotas are based purely on maximising the harvest.

See also *Species management – Legally protect reptile species* for studies that discuss comparisons of where harvesting is prohibited compared to allowed.

Auliya M. (2010) *The conservation status and impacts of trade on the Oriental Rat Snake Ptyas mucosa in Java, Indonesia*. TRAFFIC Southeast Asia, Petaling Jaya, Selangor, Malaysia.

A replicated study in 1995 and 2004 in a community in Ostional, Costa Rica (1) found that a regulated harvest of olive ridley turtle *Lepidochelys olivacea* eggs resulted in community members reporting a willingness to do more to protect turtles. A majority of survey respondents reported a willingness to do more to protect sea turtles in 1995 (67%) and in 2004 (78%). A long running programme (over 20 years at time of publication) of legalized turtle egg harvesting was established and run by a community cooperative. The project made use of the "arribada": a phenomenon of mass nesting by olive ridley turtles on nesting beaches. Members could harvest and sell turtle eggs and also carry out a range of activities relating to turtle protection, including beach cleaning, guarding and 'liberating' hatchlings (details of this not provided). A survey of households was carried out in 1995 (76 households) and followed up in 2004 (60 households).

A before-and-after study in 1975–2009 in 12 tidal rivers in the Northern Territory, Australia (2) found that after introducing regulated egg harvests and legal protection, saltwater crocodiles *Crocodylus porosus* increased in population density and average crocodile size increased over time. After saltwater crocodile harvests were regulated, relative population density of crocodiles (excluding hatchling crocodiles <0.6 m in length) increased by >three times (2009 estimate: 5 crocodiles/km; 1975 estimate: 2 crocodiles/km). The proportion of larger crocodiles (>1.8 m in length) increased over time in all rivers (most common size in 2007–2008: 2.7 m long, and in 1978–1979: 1.5 m long). Saltwater crocodiles were legally protected in the Northern Territory in 1971. Harvest of non-hatchling crocodiles was limited to <200/year and commercial fishing was banned on most rivers. A managed egg harvest was introduced in 1984–2009 (harvests in 1983–1986: 994–3,470 eggs, increasing to <50,000 in 2009–2010, see original paper for details). Saltwater crocodiles were surveyed in 12 large tidal rivers using a standardized approach (spotlight surveys at night by boat) in June–October in 1975–2009 (11–29 survey years/river, 33–138 km long surveys/river, 682 km total river length surveyed). Crocodile size was estimated when possible and only crocodiles >0.6 m ('non-hatchlings') were reported. Relative non-hatchling crocodile population densities were estimated using the sightings data divided by the length of river surveyed.

A study in 1999 and 2005–2006 in Indonesia (3) reported that regulating reptile harvests through quotas did not limit the number of tokay geckos *Gekko gecko*, Javan filesnakes *Acrochordus javanicus* and Asiatic softshell turtles *Amyda cartilaginea* that were harvested and exported. Trade in tokay geckos was estimated at 1.2 million individuals/year, compared to an annual quota of 50,000;

trade in Javan filesnakes was estimated at 330,000 individuals/year, compared to a quota of 200,000; and trade in Asiatic softshell turtles was estimated at 200,000–450,000 in 1998 and 1999 in three cities, compared to a national quota of 10,000. The Indonesian authorities set annual quotas for the harvest and export of reptile species that were not legally protected, and determined quota numbers through consultation with various stakeholders, including reptile traders. Data on trade were collected from the Indonesian authorities (CITES Management Authority), as well as through interviews with members of reptile and amphibian trade associations and other stakeholders in the reptile trade. In 1999, trade data for Asiatic softshell turtles was collected from reptile traders in three cities in Sumatra. In 2005–2006, trade data for tokay geckos was collected at four locations in Java, and Javan filesnake data was collected in five cities and involved all major exporters in the country.

A before-and-after study in 1975–2015 in two island groups in Ogasawara, Japan (4) found that in the years following regulations to limit the annual harvest of green turtles *Chelonia mydas* and a long-term programme of allowing harvested females to lay eggs before being killed the estimated number of nesting female turtles and hatchlings tended to be higher. Results were not statistically tested, and the effects of the interventions cannot be separated. The estimated number of nesting female turtles tended to be higher following regulations (180–580 turtles/year) compared to before regulation (25–210 turtles/year) and in years that regulation started (110–205 turtles/year). The number of hatchlings produced in natural nests was also higher in years after regulations were put in place (10,000–95,000 hatchlings/year) than before (0–16,000 hatchling/year). Fisheries regulations implemented in 1994 and 1997 limited the annual catch to 150 and then 135 turtles/year respectively. In 1975–2008, harvested female turtles were taken to an enclosed beach to lay multiple clutches of eggs before being killed. Surveys were conducted in May–September 1975–2015 (Chichi-jima islands) and 1988–2015 (Haha-jim islands) and used the number of nests to estimate abundance of females (see paper for details). In July–November, nests were excavated, and hatchling numbers were estimated by counting empty shells.

(1) Campbell L.M., Haalboom B.J. & Trow J. (2007) Sustainability of community-based conservation: sea turtle egg harvesting in Ostional (Costa Rica) ten years later. *Environmental Conservation*, 34, 122–131.
(2) Fukuda Y., Webb G., Manolis C., Delaney R., Letnic M., Lindner G. & Whitehead P. (2011) Recovery of saltwater crocodiles following unregulated hunting in tidal rivers of the Northern Territory, Australia. *The Journal of Wildlife Management*, 75, 1253–1266.
(3) Nijman V., Shepherd C.R. & Sanders K.L. (2012) Over-exploitation and illegal trade of reptiles in Indonesia. *The Herpetological Journal*, 22, 83–89.
(4) Kondo S., Morimoto Y., Sato T. & Suganuma H. (2017) Factors affecting the long-term population dynamics of green turtles (*Chelonia mydas*) in Ogasawara, Japan: Influence of natural and artificial production of hatchlings and harvest pressure. *Chelonian Conservation and Biology*, 16, 83–92.

6.2. Commercially breed reptiles to reduce pressure on wild populations

- **One study** evaluated the effects on reptile populations of commercially breeding reptiles to reduce pressure on wild populations. This study was in the Cayman Islands[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (0 STUDIES)

OTHER (1 STUDY)

- **Human behaviour change (1 study):** One study in the Cayman Islands[1] found that where there was a commercial turtle farm, consumption and purchase of wild turtle products was rare, though some residents still showed a preference for wild turtle meat.

Background

Wildlife breeding farms are sometimes used with the goal of alleviating the pressure of harvesting or collecting on wild reptile populations. However, if not carefully regulated, breeding farms may be used to 'launder' illegally harvested reptiles, thereby allowing illegal trade to continue (Lyons & Natusch 2011).

To be included, ideally studies should have tested the impact of commercial breeding on wild populations, and this should be taken into account when considering the effectiveness of this action. However, often the effects on wild populations are not explicitly or directly tested.

For studies that assess the impact of releasing captive-bred or head-started reptiles into the wild, see *Species management – Release captive-bred reptiles into the wild* and *Head-start wild-caught reptiles for release*.

Lyons, J.A. & Natusch D.J. (2011) Wildlife laundering through breeding farms: illegal harvest, population declines and a means of regulating the trade of green pythons (*Morelia viridis*) from Indonesia. *Biological Conservation*, 144, 3073–3081.

A study in 2014 in the Cayman Islands (1) found that where there was a commercial turtle farm, consumption and purchase of wild turtle products was rare, though some residents still showed a preference for wild turtle meat. Overall, around 1% of households illegally consumed eggs in the prior 12 months and 2% illegally bought turtle meat. Among consumers who preferred buying uncooked turtle meat, 14% showed a preference for wild meat over farmed meat. Of residents that consumed turtle during the prior 12 months, 37% bought it from the turtle farm and 62% did not buy uncooked turtle meat (e.g. consumed at restaurants). During the 12 months of the study, no source of legal, wild turtle meat was available to consumers. In 1968, a commercial breeding operation was established to provide turtle meat for consumption and reduce pressure on wild stocks. In 2014, surveys of 100 households from each of six districts were carried out, and respondents were asked about turtle meat consumption, purchase and participation in illegal behaviours relating to sea turtles (see paper for details of questioning methods). In addition, 182 consumers of turtle meat were asked further questions about their preferences.

(1) Nuno A., Blumenthal J.M., Austin T.J., Bothwell J., Ebanks-Petrie G., Godley B.J. & Broderick A.C. (2018) Understanding implications of consumer behavior for wildlife farming and sustainable wildlife trade. *Conservation Biology*, 32, 390–400.

6.3. Enforce regulations to prevent trafficking and trade of reptiles

- We found no studies that evaluated the effects on reptile populations of enforcing regulations to prevent trafficking and trade of reptiles.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Some reptile species threatened by trade are protected under the CITES agreement (Convention on International Trade in Endangered Species of Wild Fauna and Flora), which aims to regulate the international trade of endangered species. However, it is the responsibility of each participating country to adopt its own national legislation to ensure the regulations are implemented, and in some countries illegal trade continues. Alongside enforcement by exporting countries, importing countries may have an important role to play in developing solutions to mitigate the threat of trafficking and trade in reptiles.

6.4. Patrol or monitor nesting beaches

- **Seven studies** evaluated the effects of patrolling or monitoring nesting beaches on reptile populations. Three studies were in Costa Rica[2,4,6] and one was in each of the US Virgin Islands[1], Mexico[3], Mozambique[5] and the Dominican Republic[7].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Reproductive success (2 studies):** One before-and-after site comparison study in Costa Rica[6] found that olive ridley turtle nests that were moved to a patrolled hatchery and nests that were camouflaged on the nesting beach had similar hatching success. One replicated, controlled study in the Dominican Republic[7] found that on beaches with regular patrols, hatching success of leatherback turtle nests was higher than in nests relocated to hatcheries.

BEHAVIOUR (0 STUDIES)

OTHER (6 STUDIES)

- **Human behaviour change (6 studies):** Two studies in the US Virgin Islands[1] and Costa Rica[4] found that during years when beach patrols were carried out poaching of leatherback turtle nests decreased. Three studies (including two before-and-after studies) in Costa Rica[2,6] and Mexico[3] found that when beach patrols were carried out in combination with either an education programme for local communities[2], limiting beach access[3] or camouflaging nests and moving nests to a hatchery[6], poaching of leatherback turtle nests[2] and olive ridley turtle nests[3,6] decreased. One before-and-after study in Mozambique[5] found that during a community-based turtle monitoring project no green turtle egg collection or hunting of adults was recorded.

Background

Female sea turtles are vulnerable to poaching during the nesting season. Human presence on beaches at night can prevent female sea turtles from emerging from the sea to nest or interrupt females while egg laying so that they return to sea. Once laid, nests are vulnerable to human disturbance and illegal egg collection. Patrols may be carried out at frequent intervals specifically to deter human activities. Scientific monitoring programmes may function as a similar deterrent through the regular presence of officials on a beach.

Many studies report anecdotal evidence of reduced poaching while beaches are being patrolled, but few studies actually test the effect of patrols formally.

See also: *Threat: Human intrusions and disturbance – Use nest covers to protect against human disturbance*.

A study in 1981–1994 on a sandy beach in St Croix, US Virgin Islands (1) reported that when nightly beach patrols were carried out, incidents of poaching of leatherback turtle *Dermochelys coriacea* nests declined. Results were not statistically tested. In 1981 when patrolling began, incidents of poaching were highest (11%); then ranged from 0–2% in 1982–1985; then remained at 0% from 1986–1994. The authors reported anecdotes that before the study began, poaching of nests approached 100% annually (no data presented). In 1981–1994 the beach was patrolled hourly between 20:00–05:00 h every night from 1 April until no new nests had been discovered for 10 days. In 1982–1994, all nests in erosion-prone areas were relocated to stable parts of the beach immediately after laying.

A study in 1991–1992 on a sandy beach in Guanacaste Province, Costa Rica (2) found that while a combination of periodic beach patrols for turtle nest protection; beach patrols for research; and education programmes with local communities were taking place, there was a decrease in the percentage of leatherback turtle *Dermochelys coriacea* nests lost to poaching. Results were not statistically tested, and the effect of the different actions cannot be separated. The percentage of nests lost to poaching was 91% (49 of 54 nests) in October when patrols began; 51% (102 of 199 nests) in November; and 0–2% (of around 500 nests) in December–March. The beach was patrolled nightly for research purposes from October 1991–March 1992. Additional patrols were carried out by rural guards for three weeks in November and December, and periodically during January and February. In October–November 1991, an education and communications programme was carried out with local communities that involved organising trips to see the turtles, the chance to help with turtle research, lectures, lessons, slideshows, and local distribution of a brochure on leatherback turtle biology and conservation. Activities were also carried out with scout groups and the National Museum of Costa Rica (dates not provided).

A before-and-after study in 1988–1997 on a beach in Playa Cuixmala, Mexico (3) found that after limiting human access to the beach and introducing patrols, along with moving nests to an on-beach hatchery, numbers of olive ridley *Lepidochelys olivacea* nests poached were lower. Results were not statistically tested. After limiting human access to the beach and introducing regular nightly beach patrols during the nesting season, two of 2,335 olive ridley turtle nests were poached in five years, compared to >90% of 59 nests poached in the two years prior to protections being introduced. A 3 km long beach was controlled by blocking human access and conducting night patrols at 3-hour intervals during the nesting season (July–March) in 1990–1997. At the same time, a proportion of nests were collected and transported to beach hatchery. Prior to this, nesting activity and poaching was monitored on the beach in 1988–1989.

A study in 1990–2004 on one sandy beach on the Caribbean coast of Costa Rica (4) found that patrolling beaches resulted in a decline in poaching of leatherback turtle *Dermochelys coriacea* nests. Results were not statistically tested. Incidents of poaching declined over the 14-year period when beaches were patrolled, from 55% of nests poached in 1990, to 13% in 1995, 9% in 2000 and 1% in 2004. The authors reported that most poaching events took place close to public access points to the beach. In February–July 1990–2004, the beach was patrolled every night for a total of 8 h (20:00–04:00 h). The main purpose of patrols was to locate nesting female turtles and to relocate nests laid in high-risk areas to an on-beach hatchery.

A before-and-after study in 2003–2007 on three beaches on Vamizi Island, Mozambique (5) found that a community-based turtle monitoring project appeared to reduce egg collection and hunting of adult green turtles *Chelonia mydas*. During the four years of a community turtle monitoring project, no egg collection (122 nests were laid/year on average) or hunting of female turtles was recorded. The authors reported that prior to the turtle monitoring project beginning, egg collection and hunting of adult female turtles was common within the local fishing community. Following the formation of two fishing village committees to manage local fishing resources and implement regulations, the committees created a turtle sanctuary around the north-east of the island to protect turtle breeding and feeding grounds. Three nesting beaches were monitored nightly for several months/year by 15 local turtle monitors supervised by a marine biologist in January–July 2003–2007.

A before-and-after, site comparison study in 2005–2012 on a beach in Costa Rica (6) found that relocating olive ridley *Lepidochelys olivacea* turtle nests to an on-beach hatchery with 24-hour monitoring or camouflaging them on the nesting beach tended to lead to similar hatching rates and lower egg poaching rates. Results were not statistically tested. In total, 79% of nests relocated to the hatchery and of nests camouflaged on the beach successfully hatched. Egg poaching reduced from 85% in 2005 to 10% of eggs in 2005–2012. The emergence rate of hatchlings from hatchery nests was 77%, compared to 71% of hatchlings from camouflaged nests. Nesting activity was monitored by nightly beach patrols (4x 4 hours/night) in July/August–December in 2006–2012 (958 nests were laid, 98–177/year). Nests were either relocated to an on-beach hatchery (363 nests, 38%), or camouflaged (595 nests, 61%) to discourage illegal collecting. Relocated nests were randomly allocated a 1 m^2 plot in the hatchery and dug into the sand. The hatchery was monitored 24 hours a day during the nesting season. Hatchlings were monitored on emergence and nests were excavated after hatching due dates to check hatching success.

A replicated, controlled study in 2008–2009 on five sandy beaches in southwest Dominican Republic (7) found that when beaches were patrolled regularly during the nesting season, leatherback turtle *Dermochelys coriacea* nest hatching success was higher than when nests were relocated for artificial incubation. Results were not statistically tested. Over two years, hatching success of nests left in situ on a beach with limited human access and regular night patrols was 74–85% compared to 34–58% for nests moved to hatcheries. Eggs were relocated from five beaches in a national park (1,374 km^2). In March–August 2008–2009, the beaches were surveyed during the day and night for signs of nesting (daily – at least every other week) and nests were relocated. On beaches with high pressure of illegal collecting, 35 nests (all nests found) were relocated. On beaches with more limited human access, 31 nests were relocated and 43 were left in situ and monitored to hatching. Eggs from relocated nests were placed with sand in polystyrene boxes and moved to wooden huts near the nesting beaches. On beaches where nests were left in situ, nightly patrols were carried out by government rangers 2–3 nights/week in April–May. Beaches with all nests relocated were also patrolled regularly at night, but as all nests were removed from these beaches, no in situ results were reported.

(1) Boulon Jr R.H., Dutton P.H. & McDonald D.L. (1996) Leatherback Turtles (*Dermochelys coriacea*) on St. Croix, U. S. Virgin Islands: Fifteen Years of Conservation. *Chelonian Conservation and Biology*, 2, 141–147.
(2) Chaves-Quirós A.C., Serrano G., Marín G., Arguedas-Campos E., Jimenez A. & Spotila J.R. (1996) Biology and conservation of leatherback turtles, *Dermochelys coriacea*, at Playa Langosta, Costa Rica. Biología y conservación de las tortugas baulas, *Dermochelys coriacea*, en Playa Langosta, Costa Rica. *Chelonian Conservation and Biology*, 2, 184–189.
(3) García A., Ceballos G. & Adaya R. (2003) Intensive beach management as an improved sea turtle conservation strategy in Mexico. *Biological Conservation*, 111, 253–261.
(4) Chacón-Chaverri D. & Eckert K.L. (2007) Leatherback sea turtle nesting at Gandoca Beach in Caribbean Costa Rica: management recommendations from fifteen years of conservation. *Chelonian Conservation and Biology*, 6, 101–110.
(5) Garnier J., Hill N., Guissamulo A., Silva I., Witt M. & Godley B. (2012) Status and community-based conservation of marine turtles in the northern Querimbas Islands (Mozambique). *Oryx*, 46, 359–367.
(6) James R. & Melero D. (2015) Nesting and conservation of the Olive Ridley sea turtle (*Lepidochelys olivacea*) in playa Drake, Osa Peninsula, Costa Rica (2006–2012). *Revista De Biologia Tropical*, 63, 117–129.
(7) Revuelta O., León Y.M., Broderick A.C., Feliz P., Godley B.J., Balbuena J.A., Mason A., Poulton K., Savoré S., Raga J.A., Tomás J. (2015) Assessing the efficacy of direct conservation interventions: clutch protection of the leatherback marine turtle in the Dominican Republic. *Oryx*, 49, 677–686.

6.5. Introduce alternative income sources to replace hunting or harvesting of reptiles

- **One study** evaluated the effects on reptile populations of introducing alternative income sources to replace hunting or harvesting of reptile populations. This study was in St Kitts[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (0 STUDIES)

OTHER (1 STUDY)

- **Human behaviour change (1 study):** One before-and-after study in St Kitts[1] found that fishers that took jobs on a turtle management project reported that they ceased turtle fishing activity.

Background
Introducing appropriate alternative income-generating activities may reduce the reliance of communities on earning money from collecting/hunting and selling wild animals, and in turn reduce the collecting/hunting pressure on wild populations. Studies in this section should ideally directly test the effect of the intervention on wild populations, as well as the income benefits and participation levels within the communities.

A before-and-after study in 2006–2009 in St Kitts (1) found that offering alternative livelihoods relating to sea turtle management to sea turtle fishers resulted in fishers reporting that they had ceased turtle fishing activity. Fishers that accepted jobs on the turtle management project reported that they had stopped harvesting sea turtles as a result. Prior to this, fishers reported that they caught between 25 and 100 turtles/year. In 2006, an initial survey of seven fishers was carried out that assessed how dependent fishers were on the sea turtles and how many they were capturing. Fishers that expressed interest in a Fishers' Technician Programme were offered positions on the turtle management project. Those fishers that took up positions on the technician programme subsequently reported on their sea turtle harvesting activities (details of reporting method are unclear).

(1) Stewart K.M., Norton T.M., Tackes D.S. & Mitchell M.A. (2016) Leatherback ecotourism development, implementation, and outcome assessment in St. Kitts, West Indies. *Chelonian Conservation and Biology*, 15, 197–205.

Reduce unwanted catch

Spatial and temporal management

6.6. Cease or prohibit all types of fishing

- We found no studies that evaluated the effects on reptile populations of ceasing or prohibiting all types of fishing.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background
Fishing can impact reptile populations directly by species removal (both intentional and unintentional), or indirectly through removal of other species or by damaging habitats. Ceasing fishing activities in specific areas may allow time for populations of reptiles and the species they rely on (e.g. prey species) to recover.

For studies that investigate the effect of ceasing or prohibiting particular types of fishing see *Cease or prohibit commercial fishing* and *Limit or prohibit specific fishing methods*. For studies that investigate the effect of ceasing or prohibiting fishing at certain times see *Establish temporary fishery closures*.

6.7. Cease or prohibit commercial fishing

- We found no studies that evaluated the effects on reptile populations of ceasing or prohibiting commercial fishing.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Commercial fishing is extraction of marine organisms by any method for sale and profit and poses a threat to reptiles through accidental entanglements and unwanted catch ('bycatch'). Ceasing or prohibiting commercial fishing in specific areas may allow time for populations of reptiles and the species they rely on (e.g. prey species) to recover.

For other studies that investigate the effect of ceasing or prohibiting fishing see *Cease or prohibit all types of fishing* and *Limit or prohibit specific fishing methods*. For studies that investigate the effect of ceasing or prohibiting fishing at certain times see *Establish temporary fishery closures*.

6.8. Establish temporary fishery closures

- **Three studies** evaluated the effects of establishing temporary fishery closures on reptile populations. Two studies were in the USA[1,2] and one was in Brazil[3].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (3 STUDIES)

- **Abundance (1 study):** One site comparison study in Brazil[3] found that areas where a fishing agreement was implemented that involved seasonal fishing restrictions along with a wider set of measures had more river turtles than areas that did not implement the agreement.
- **Survival (2 studies):** One replicated, before-and-after study in the USA[1] found that during seasonal closures of shrimp trawling there were fewer lethal strandings of loggerhead and Kemp's ridley turtles. One study in the USA[2] found that following the re-opening of a swordfish long-line fishery with turtle catch limits in place, loggerhead turtle bycatch reached the annual catch limit in two of three years, and when the limit was reached the fishery was closed for the rest of the year.

BEHAVIOUR (0 STUDIES)

Background

Establishing temporary fishery closures in an area can remove the direct risk of reptiles being caught or entangled in fishing gear for the designated period. Examples of temporary closures might include seasonal closures, for example undertaken to protect turtles in foraging grounds, or move-on rules whereby temporary closure of a fished area occurs when a catch or by-catch threshold is reached (e.g. Dunn *et al.* 2014).

For other studies that investigate the effect of ceasing or prohibiting fishing see *Cease or prohibit all types of fishing*; *Cease or prohibit commercial fishing* and *Limit or prohibit specific fishing methods*.

Dunn D.C., Boustany A.M., Roberts J.J., Brazer E., Sanderson M., Gardner B. & Halpin P.N. (2014) Empirical move-on rules to inform fishing strategies: a New England case study. *Fish and Fisheries*, 15, 359–375.

A replicated, before-and-after study in 1980–2000 in two coastal areas in the Gulf of Mexico, Texas, USA (1) found that seasonal area closures to shrimp trawling in nearshore waters reduced lethal strandings of loggerhead *Caretta caretta* and Kemp’s ridley *Lepidochelys kempii* turtles. One statutory closure reduced lethal strandings of both loggerhead and Kemp’s ridley turtles for the 6–8-week duration of the closure compared to when it is not in effect (data reported as model outputs). When the second statutory closure was in effect, 6–8 Kemp’s ridley turtles were stranded inside the closed area, compared to the 13 turtles in the year prior to the closure taking effect (results were not statistically tested). Two statutory closures were implemented to restrict shrimp trawling within designated distances of Texas shores. The first excluded shrimping from all Texan Gulf of Mexico shores to 200 nm in 15 May–15 July each year (dates variable based on shrimp stocks; effective from 1981, updated in 1990). The second prohibited shrimp fishing within five miles of Padre Island on the south Texas coast from 1 December–15 July, effective from December 2000. Data from the Sea Turtle Stranding and Salvage Network (1980–2000) were used to analyse the effect of closures. Incidental turtle catch, captive-reared/head-started turtles and turtles below <10 cm were excluded from analysis.

A study in 2005–2007 in pelagic waters north of Hawaii, USA (2) found that after annual catch limits were established for loggerhead turtles *Caretta caretta* in a swordfish *Xiphias gladius* shallow-set longline fishery, turtle catch reached the limit in the second year after the fishery re-opened with catch limits but was lower in the first and third year. Results were not statistically tested. In the first year after the fishery re-opened with a turtle catch limit, nine loggerhead turtles (0.004–0.049 turtles/1,000 hooks) were caught, but in the second year the catch limit of 17 turtles (0.013–0.044 turtles/1000 hooks) was reached and the fishery was closed for the rest of the year. In the third year, 12 turtles were caught (0.0–0.028 turtles/1,000 hooks). In late 2004, the fishery re-opened after a two-year shut down due the high number of loggerhead turtle catch levels. After re-opening, a catch limit of 17 turtles/year was established, after which the fishery would close for the rest of the year. In January–March 2005–2007, line deployments (2005: 520; 2006: 842; 2007: 797), hooks put out (2005: 429,580; 2006: 670,914; 2007: 689,486), and loggerhead turtle interactions were monitored. In January–March 2007, fishers were also provided daily information in electronic and paper format from a 'TurtleWatch' tool that recommended areas to avoid to reduce turtle interactions (see original paper).

A site comparison study in 2009 on a flood plain with a variety of lakes and channels in Pará, Brazil (3) found that areas that had community-based management (CBM) of fishing practices – including seasonal fishing restrictions, limiting use of gill-nets, protecting turtle nesting beaches and a ban on turtle trading – had more river turtles *Podocnemis sextuberculata*, *Podocnemis unifilis* and *Podocnemis expansa* than areas without CBM. The effect of different aspects of the management programme cannot be separated. Turtles were more abundant in areas with CBM (321 individuals) than in areas without CBM (33 individuals). For *Podocnemis sextuberculata*, abundance was higher in areas with CBM (14 individuals/1,000 m^2 netting/12 hours) than in areas without (2 individuals/1,000 m^2 netting/12 hours), and turtle biomass was also greater (with CBM: 20 kg/1,000 m^2 netting/12 hours; no CBM: 3 kg/1,000 m^2 netting/12 hours). The fishing agreement that formed the CBM programme had been in place for 20–30 years. While 13 communities in the area were a part of the fishing agreement, only two implemented the agreement. Turtle numbers were sampled at 14 sites (7 with CBM; 7 without CBM) in August–October 2009 using gill nets (15 nets/site; 215 m^2 nets; 3 each of 5 mesh sizes) with help from local fishers.

(1) Lewison R.L., Crowder L.B. & Shaver D.J. (2003) The impact of turtle excluder devices and fisheries closures on loggerhead and Kemp's ridley strandings in the Western Gulf of Mexico. *Conservation Biology*, 17, 1089–1097.
(2) Howell E.A., Kobayashi D.R., Parker D.M., Balazs G.H. & Polovina J.J. (2008) TurtleWatch: A tool to aid in the bycatch reduction of loggerhead turtles *Caretta caretta* in the Hawaii-based pelagic longline fishery. *Endangered Species Research*, 5, 267–278.
(3) Miorando P.S., Rebêlo G.H., Pignati M.T. & Brito Pezzuti J.C. (2013) Effects of community-based management on Amazon river turtles: a case study of Podocnemis sextuberculata in the lower Amazon floodplain, Pará, Brazil. *Chelonian Conservation and Biology*, 12, 143–150.

6.9. Limit or prohibit specific fishing methods

- **One study** evaluated the effects of limiting or prohibiting specific fishing methods on reptile populations. This study was in Brazil[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Abundance (1 study):** One site comparison study in Brazil[1] found that in areas where a fishing agreement was implemented that involved limiting the use of gill nets along with a wider suit of measures had more river turtles than areas that did not implement the agreement.

BEHAVIOUR (0 STUDIES)

Background

Fishing can impact reptiles by catching individuals unintentionally or causing injury or death through entanglements in fishing gear. Applying restrictions to the use of specific fishing methods (which may include 'static' methods such as using lobster/crab pots and traps, or 'mobile' methods such as gill nets or trawl nets) in locations or seasons when reptiles are particularly vulnerable to capture may alleviate these threats.

For other studies that investigate the effect of ceasing or prohibiting fishing see *Cease or prohibit all types of fishing* and *Cease or prohibit commercial fishing*. For studies that investigate the effect of ceasing or prohibiting fishing at certain times see *Establish temporary fishery closures*.

A site comparison study in 2009 on a flood plain with a variety of lakes and channels in Pará, Brazil (1) found that areas that had community-based management (CBM) of fishing practices – including limiting use of gill-nets, seasonal fishing restrictions, protecting turtle nesting beaches and a ban on turtle trading – had more river turtles *Podocnemis sextuberculata*, *Podocnemis unifilis* and *Podocnemis expansa* than areas without CBM. The effect of different aspects of the management programme cannot be separated. Turtles were more abundant in areas with CBM (321 individuals) than in areas without CBM (33 individuals). For *Podocnemis sextuberculata*, abundance was higher in areas with CBM (14 individuals/1,000 m^2 netting/12 hours) than in areas without (2 individuals/1,000 m^2 netting/12 hours), and turtle biomass was also greater (with CBM: 20 kg/1,000 m^2 netting/12 hours; without CBM: 3 kg/1,000 m^2 netting/12 hours). The fishing agreement that formed the CBM programme had been in place for 20–30 years. While 13 communities in the area were a part of the fishing agreement, only two implemented the agreement. Turtle numbers were sampled at 14 sites (7 with CBM; 7 without CBM) in August–October 2009 using gill nets (15 nets/site; 215 m^2 nets; 3 each of 5 mesh sizes) with help from local fishers.

(1) Miorando P.S., Rebêlo G.H., Pignati M.T. & Brito Pezzuti J.C. (2013) Effects of community-based management on Amazon river turtles: a case study of *Podocnemis sextuberculata* in the lower Amazon floodplain, Pará, Brazil. *Chelonian Conservation and Biology*, 12, 143–150.

6.10. Deploy fishing gear at different depths

- **Three studies** evaluated the effects of deploying fishing gear at different depths on reptile populations. One study was in each of Canada[1], off the coast of Mexico[2] and the Atlantic[3].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Survival (1 study):** One replicated, randomized, paired study in Canada[1] found that no turtles died in floated nets, but some died in submerged nets.
- **Condition (1 study):** One replicated, randomized, paired study in Canada[1] found that turtles caught in floated nets were less at risk of drowning than those caught in submerged nets.

BEHAVIOUR (0 STUDIES)

OTHER (3 STUDIES)

- **Unwanted catch (3 studies):** Two of three studies (including two replicated studies) in Canada[1], Mexico[2] and the Atlantic[3] found that bottom-set fishing nets with fewer buoys caught fewer sea turtles than standard nets[2] or that fewer loggerhead turtles were caught when longline hooks were set below 22 m deep, but the number of leatherback turtles caught was unaffected by hook depth[3]. The other study[1] found that floated and submerged nets caught a similar number of turtle species.

Background

Deploying fishing gear at different depths may reduce interactions with aquatic reptiles and subsequent entanglements and unwanted catch ('bycatch'). For example, nets may be partially submerged and allow access to the surface, or lines may be set at depths outside of the foraging range of key species. However, the feasibility of this intervention will depend on the type of fishery and the ecology of the target species.

See also: *Modify fishing gear to reduce mortality in the event of unwanted catch*.

A replicated, randomized, paired study in 2009–2010 in a freshwater lake in Ontario, Canada (1) found that using floated nets did not reduce levels of unwanted catch but did reduce drowning risk (measured using blood lactate levels) and mortality in turtles caught in fyke nets. Turtle catch rates and species composition were similar between floated (0.06 turtles/hour, 35 individuals) and submerged nets (0.10 turtles/hour, 48 individuals). Blood lactate levels (a measure of drowning risk) were reduced in turtles tested in floated nets (1.3–3.5 mmol/L) compared to submerged nets (13.2–16.4 mmol/L). Turtle mortality only occurred in submerged nets (3 individuals, no statistical tests were carried out). Species composition was similar between net types (data presented as statistical model outputs). Turtle species caught included painted turtles *Chrysemys picta*, eastern musk turtles *Sternotherus odoratus*, northern map turtles *Graptemys geographica*, and common snapping turtle *Chelydra serpentina*. Target fish catch rates were similar between floated (2.5 fish/hour) and submerged nets (3.1 fish/hour). In August 2009, experimental tests of blood lactate and pH levels of turtles placed in submerged, floated and semi-submerged nets were carried out initially to test whether turtles used air spaces provided by elevating nets in the water (9–10 trials/net, see original paper for details). In April–June 2010, submerged nets (without floats) and nets with floats were deployed in pairs (two submerged nets deployed within 15 m of two floated nets) for 8–48 hours in 30 locations (1–2 m deep). Blood samples were taken from all turtles and the number caught (including mortalities) was recorded.

A controlled study in 2007–2009 on the sea floor in Baja California Sur, Mexico (2) found that reducing the number of buoys attached to bottom-set fishing nets reduced unwanted catch of sea turtles. Reduced-buoy nets caught fewer sea turtles (0.06 turtles/100 m of net/day) compared to standard nets (0.19 turtles/100 m of net/day). Unwanted catch included loggerhead *Caretta caretta*, green turtle *Chelonia mydas* and olive ridley turtles *Lepidochelys olivacea*. Average catch of target fish species was similar in both net types (reduced-buoy: 10; standard: 12 kg/100 m of net/day) although the market value of target fish was lower in reduced-buoy nets ($18/trip) compared to standard nets ($25/trip). Reduced buoy nets (1 buoy/8.5 m net) and standard nets (1 buoy/1.7 m net; both net types were 111–120 m long and 4–6 m high) were deployed in pairs for 21–25 hours at a time during summer 2007 (40 deployments), 2008 (40 deployments) and 2009 (96 deployments). The market value of target catch species was calculated based on the catch composition.

A replicated study in 1992–2015 in pelagic longline fisheries in the Atlantic (3) found that longlines with deeper hooks caught fewer loggerhead turtles *Caretta caretta*, but bycatch of leatherbacks *Dermochelys coriacea* was not affected by hook depth. The chance of catching loggerheads was lower when hooks were around 22 m deep or more (data presented as statistical model results), but leatherback catch was unaffected by hook depth. Maximum hook depth was calculated by adding up the length of the floatline, branchline and dropline. Pelagic Observer Program data from (1992–2015) was used to determine the number of turtles caught/1,000 hooks. Variation in practices relating to hook depth was used to test its effect on bycatch.

(1) Larocque S.M., Cooke S.J. & Blouin-Demers G. (2012) A breath of fresh air: avoiding anoxia and mortality of freshwater turtles in fyke nets by the use of floats. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 22, 198–205.
(2) Peckham S.H., Lucero-Romero J., Maldonado-Díaz D., Rodríguez-Sánchez A., Senko J., Wojakowski M. & Gaos A. (2016) Buoyless nets reduce sea turtle bycatch in coastal net fisheries. *Conservation Letters*, 9, 114–121.
(3) Swimmer Y., Gutierrez A., Bigelow K., Barceló C., Schroeder B., Keene K., Shattenkirk K. & Foster D.G. (2017) Sea turtle bycatch mitigation in U.S. longline fisheries. *Frontiers in Marine Science*, 4, 260.

Capacity controls

6.11. Set commercial catch quotas

- We found no studies that evaluated the effects on reptile populations of setting commercial catch quotas.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Commercial fishing and harvest quotas (such as 'Total Allowable Catch') are a means by which many governments and local regulatory bodies regulate biological resources i.e. species stocks. Setting catch quotas for specific fisheries (for instance cod), can potentially reduce the pressure on other species not targeted by the fishery but commonly affected or caught during fishing operations.

See also: *Set unwanted catch quotas*.

6.12. Set unwanted catch quotas

- **One study** evaluated the effects of setting unwanted catch quotas on reptile populations. This study was in the USA[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Survival (1 study):** One study in the USA[1] found that following the re-opening of a swordfish long-line fishery with turtle catch limits in place, loggerhead turtle bycatch reached the annual catch limit in two of three years, and when the limit was reached the fishery was closed for the rest of the year.

BEHAVIOUR (0 STUDIES)

Background

Fishing can impact reptiles by catching individuals unintentionally or causing injury or death through entanglements in fishing gear. Unwanted catch ('bycatch') quotas are used to set catch limits for unwanted species. When the quota for a particular species is reached, the fishery may be closed to all forms of fishing likely to catch that species. This may reduce the pressure on populations of reptiles that are caught accidentally and allow time for populations to recover.

See also: *Set commercial catch quotas*.

A study in 2005–2007 in pelagic waters north of Hawaii, USA (1) found that after an annual unwanted catch quota was established for loggerhead turtles *Caretta caretta* in a swordfish *Xiphias gladius* shallow-set longline fishery, turtle catch reached the limit in the second year after the fishery re-opened with catch limits, but was lower in the first and third year. Results were not statistically tested. In the first year after the fishery re-opened with a turtle catch limit, nine loggerhead turtles (0.004–0.049 turtles/1,000 hooks) were caught, but in the second year the catch limit of 17 turtles (0.013–0.044 turtles/1000 hooks) was reached and the fishery was closed for the rest of the year. In the third year, 12 turtles were caught (0.0–0.028 turtles/1,000 hooks). In late 2004, the fishery re-opened after a two-year shut down due the high number of loggerhead turtle catch levels. After re-opening, a catch limit of 17 turtles/year was established, after which the fishery would close for the rest of the year. In January–March 2005–2007, line deployments (2005: 520; 2006: 842; 2007: 797), hooks put out (2005: 429,580; 2006: 670,914; 2007: 689,486), and loggerhead turtle interactions were monitored. In January–March 2007, fishers were also provided daily information in electronic and paper format from a 'TurtleWatch' tool that recommended areas to avoid to reduce turtle interactions (see original paper).

(1) Howell E.A., Kobayashi D.R., Parker D.M., Balazs G.H. & Polovina J.J. (2008) TurtleWatch: A tool to aid in the bycatch reduction of loggerhead turtles *Caretta caretta* in the Hawaii-based pelagic longline fishery. *Endangered Species Research*, 5, 267–278.

6.13. Limit the number of fishing vessels or fishing days in an area

- We found no studies that evaluated the effects on reptile populations of limiting the number of fishing vessels or fishing days in an area.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Limiting the number of fishing vessels or days in which an area can be fished may reduce the risk of entanglements and unwanted catch ('bycatch') of aquatic reptiles. This may involve the rotation of fishing areas. Careful planning may be required to ensure that fishing effort is not redirected to other areas with a high reptile density.

See also: *Limit the length of fishing gear or density of traps in an area* and *Reduce duration of time fishing gear is in the water*.

6.14. Limit the length of fishing gear or density of traps in an area

- We found no studies that evaluated the effects on reptile populations of limiting the length of fishing gear or density of traps in an area.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Limiting the length of fishing gear (e.g. ropes, lines, nets) or the number of traps in an area may reduce the risk of entanglements and unwanted catch('bycatch') of aquatic reptiles. This could involve using shorter ropes or lines, or using multiple pots, traps or nets on each line.

See also: *Limit the number of fishing vessels or fishing days in an area* and *Reduce duration of time fishing gear is in the water*.

6.15. Reduce duration of time fishing gear is in the water

- **Two studies** evaluated the effects on reptile populations of reducing the duration of time fishing gear is in the water. One study was in the Gulf of Gabès[1] (Tunisia) and one was in the Atlantic and North Pacific[2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Survival (1 study):** One randomized study in the Gulf of Gabès[1] found that retrieving longlines immediately resulted in fewer loggerhead turtles dying compared to when line retrieval was delayed.

BEHAVIOUR (0 STUDIES)

OTHER (2 STUDIES)

- **Unwanted catch (2 studies):** One randomized study in the Gulf of Gabès[1] and one replicated study in the Atlantic and North Pacific[2] found that the amount of time that longlines were in the water for did not affect the number of loggerhead turtles[1] or leatherback and loggerhead turtles[2] caught.

Background

Reducing the length of time that the gear is left in the water ('soak time') reduces the opportunity for aquatic reptiles to become entangled in fishing gear. It may also reduce mortality rates, because when reptiles become entangled they are unable to reach the surface to replenish oxygen levels and so may drown.

See also: *Limit the number of fishing vessels or fishing days in an area* and *Limit the length of fishing gear or density of traps in an area*.

A randomized study in 2007–2008 on the sea bottom in the Gulf of Gabès, Tunisia (1) found that reducing the time longlines were in the water did not reduce the rate of unwanted catch of loggerhead turtles *Caretta caretta* in a bottom longline fishery, but did reduce the mortality rate of turtles caught. A similar number of turtles were caught when lines were retrieved immediately (immediate retrieval: 0.18 turtles/1,000 hooks) or left in the water for longer periods (1–2 hour soak: 0.34 turtles/1,000 hooks; >2 hours soak: 0.49 turtles/1,000 hooks; 16 turtles in total). Mortality rates of turtles caught accidentally were lower when bottom longlines were retrieved immediately (0/3 turtles died) compared to when longlines were retrieved after 1–2 hours (2/6 turtles died) or after more than 2 hours (5/7 turtles died). Turtle data was collected by onboard observers in 38 bottom longline deployments during 20 randomly selected fishing trips (1–3 deployments/trip) in July–September 2007–2008. Longline deployments consisted of a 10–12 km longline anchored to the seabed with 1 m long branchlines with hooks (48,020 total hooks deployed). Frozen Sardinella *Sardinella aurita* or common cuttlefish *Sepia officinalis* were used as bait. Longline deployments took place at any time of day and lines were retrieved either immediately, after 1–2 hours, or after >2 hours.

A replicated study in 1992–2015 in pelagic longline fisheries in the Atlantic and North Pacific (2) found that the amount of time longlines were in the water did not affect that number of leatherback *Dermochelys coriacea* and loggerhead *Caretta caretta* turtles caught as bycatch. The amount of time lines were in the water did not affect the chance of catching turtles (data presented as statistical model results) over the range of durations surveyed (around 8–12 hours). Duration was measured as the time between the line being set and when it was hauled in and varied between around 8–12 hours. Pelagic Observer Program data from (1992–2015) was used to determine the number of turtles caught/1,000 hooks, and variation in the amount of time lines were deployed for was used to test its effect on bycatch.

(1) Echwikhi K., Jribi I., Bradai M.N. & Bouain A. (2012) Interactions of loggerhead turtle with bottom longline fishery in the Gulf of Gabès, Tunisia. *Journal of the Marine Biological Association of the United Kingdom*, 92, 853–858.

(2) Swimmer Y., Gutierrez A., Bigelow K., Barceló C., Schroeder B., Keene K., Shattenkirk K. & Foster D.G. (2017) Sea turtle bycatch mitigation in U.S. longline fisheries. *Frontiers in Marine Science*, 4, 260.

Modify fishing gear and practices

6.16. Use visual deterrents on fishing gear

- **Two studies** evaluated the effects of using visual deterrents on fishing gear on reptile populations. One study was off the coast of Mexico[1] and one was in the USA[2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (1 STUDY)

- **Behaviour change (1 study):** One replicated, controlled study in the USA[2] found that shark-shaped and spherical deterrents had mixed effects on a range of captive loggerhead turtle behaviours.

OTHER (1 STUDY)

- **Unwanted catch: (1 study):** One replicated, controlled study off the coast of Mexico[1] found that gillnets with floating shark shapes attached to them caught fewer green turtles than unmodified nets.

Background

Reptiles, particularly sea turtles, use visual cues as part of their foraging behaviour (Wang *et al*. 2009). Taking advantage of this reliance on visual cues by using visual deterrents, such as shark shapes, could be a way to discourage turtle interactions with fishing gear.

See also: *Add lights to fishing gear*.

Wang J., Fisler S. & Swimmer Y. (2009) *Developing visual deterrents to reduce sea turtle bycatch: testing shark shapes and net illumination*. Proceedings – Proceedings of the technical workshop on mitigating sea turtle bycatch in coastal net fisheries, Honolulu, USA, 49–50.

A replicated, controlled study in 2006–2009 in surface waters of a coastal lagoon and on the sea floor in the Baja California peninsula, Mexico (1) found that attaching floating shark shapes to gillnets reduced unwanted catch of green turtles *Chelonia mydas*. Shark shapes attached to nets reduced the catch of green turtles by 54% (6 turtles/12 h soak of 100 m net) compared to unmodified nets (12 turtles/12 h soak of 100 m net). Commercially-targeted fish catch was reduced in nets with shark shapes (6 fish/12 h soak of 200 m net) compared to unmodified nets (11 fish/12 h soak of 200 m net). Dark-painted cut-out shark shapes were weighted and attached every 10 m to gillnets. Shark nets were deployed in pairs with unmodified nets < 1 km away. In total, 14 trials were placed at the surface to test sea turtle catch (60–95 m gillnets, July 2006, May–September 2007–2008) and 22 trials were carried out to test fish catch rates on commercial fishing vessels in a bottom-set gillnet fishery (200–400 m nets set 200 m apart at 10–30 m depths, May-September 2009). All nets were deployed in daylight.

A replicated, controlled study (year not given) in laboratory conditions in Texas, USA (2) found that shark-shaped and spherical deterrents did not deter captive-reared loggerhead turtles *Caretta caretta* from biting squid bait, but that shark models resulted in changes in four out of five behaviours prior to biting compared to when spheres or no deterrent was used. The number of trials in which turtles did not bite the squid was similar across treatments (shark model: 11%; sphere: 9%; no deterrent: 3%). Turtles took longer to bite the squid in the presence of the shark model compared to no deterrent (shark model: 5 minutes; no deterrent: 2 minutes), but the time was statistically similar to when the sphere was used (2 minutes). Turtles approached the squid fewer times in the presence of the shark model (10 times/15 minutes) compared to the sphere (13 times) and no deterrent (16 times). Time spent away from the bait and the number of carapace turns were also affected by the deterrent, but the number of breaths taken was not (see paper for details). Forty-two captive-reared turtles (30–33 months old) were individually presented with either a shark model (91 cm long) with squid bait, a sphere (28 cm diameter) with squid bait or just a squid in three separate trials (1 turtle/trial). Turtles had a gap of three weeks between each trial and were fasted for three days before the trial started. Trials were conducted in a fibreglass tank (90 x 74 x 406 cm) with a water depth of 59 cm. Trials (15 minute acclimation and 15 minute trial) were video recorded, and the six behaviours were measured.

(1) Wang J.H., Fisler S. & Swimmer Y. (2010) Developing visual deterrents to reduce sea turtle bycatch in gill net fisheries. *Marine Ecology Progress Series*, 408, 241–250.

(2) Bostwick A., Higgins B.M., Landry Jr A.M. & McCracken M.L. (2014) Novel use of a shark model to elicit innate behavioral responses in sea turtles: Application to bycatch reduction in commercial fisheries. *Chelonian Conservation and Biology*, 13, 237–246.

6.17. Add lights to fishing gear

- **Five studies** evaluated the effects of adding lights to fishing gear on reptile populations. Two studies were in the Baja California peninsula[1a,1b] (Mexico) and one was in each of Sechura Bay[2] (Peru), the Atlantic and North Pacific[3] and the Adriatic Sea[4].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Survival (1 study):** One randomized, controlled, paired study in the Adriatic Sea[4] found that no loggerhead turtles were caught and died in in gillnets with UV lights whereas some did in nets without lights.

BEHAVIOUR (0 STUDIES)

OTHER (5 STUDIES)

- **Unwanted catch (5 studies):** Four controlled studies (including three replicated and two paired studies) in the Baja California peninsula[1a,1b], Sechura Bay[2] and the Adriatic Sea[4] found that gillnets with LED lights[1a,2], light sticks[1b] or UV lights[4] caught fewer green turtles[1a,1b,2] and loggerhead turtles[4] than nets without lights. One replicated study in the Atlantic and North Pacific[3] found mixed effects of increasing the number of light sticks on longlines on the chance of catching loggerhead and leatherback turtles.

Background

Reptiles, particularly sea turtles, use visual cues as part of their foraging behaviour (Wang *et al*. 2009). Taking advantage of this reliance on visual cues by adding lights to fishing gear could be a way to reduce turtle interactions with fishing gear.

See also: *Use visual deterrents on fishing gear*.

Wang J., Fisler S. & Swimmer Y. (2009) *Developing visual deterrents to reduce sea turtle bycatch: testing shark shapes and net illumination*. Proceedings – Proceedings of the technical workshop on mitigating sea turtle bycatch in coastal net fisheries, Honolulu, USA, 49–50.

A replicated, controlled study in 2006–2009 in surface waters of a coastal lagoon and on the sea floor in the Baja California peninsula, Mexico (1a) found that attaching LED lights to gillnets reduced unwanted catch of green turtles *Chelonia mydas*. LED-lit nets reduced turtle catch by 40% (7 turtles/12 h x 100 m net) compared to unmodified nets (12 turtles/12 h x 100 m net). Catch of commercially targeted fish was similar in LED-lit nets (11 fish/12h x 200 m net) compared to unmodified nets (11 fish/12 h x 200 m net). Green LEDs were attached every 10 m to the float line of gillnets. LED-lit gillnets were deployed in pairs < 1 km away from nets that had inactive LEDs attached (unmodified nets). In total, 15 trials were carried out at surface level to test sea turtle catch (60–95 m gillnets, July 2006, May-September 2007–2008) and 23 trials were carried out to test fish catch rates on commercial fishing vessels in a bottom-set gillnet fishery (200–400 m gillnets set 200 m apart at 10–30 m depths, May–September 2009). All nets were deployed in the dark.

A replicated, controlled study in 2006–2009 in surface waters of a coastal lagoon and on the sea floor in the Baja California peninsula, Mexico (1b) found that attaching chemical light sticks to gillnets reduced unwanted catch of green turtles *Chelonia mydas*. Light stick-lit nets reduced turtle catch by 59% (8 turtles/12 h x 100 m net) compared to unmodified nets (19 turtles/12 h x 100 m net). Catch of commercially targeted fish was similar in light stick-lit nets (12 fish/12h x 200 m net) compared to unmodified nets (13 fish/12 h x 200 m net). Green chemiluminescent light sticks (15 cm) were attached every 5 m to the float line of gillnets. Illuminated nets were deployed in pairs < 1 km away from gillnets that had inactive light sticks attached (unmodified nets). In total, six trials were carried out at surface level to test sea turtle catch (60–95 m gillnets, July 2006, May-September 2007–2008and 17 trials were carried out to test fish catch rates on commercial fishing vessels in a bottom-set gillnet fishery (200–400 m gillnets set 200 m apart at 10–30 m depths, May–September 2009). All nets were deployed in the dark.

A replicated, controlled, paired study in 2011–2013 on the seafloor in Sechura Bay, northern Peru (2) found that LED net illuminators reduced unwanted catch of green turtles *Chelonia mydas* in a bottom-set gillnet fishery. Green turtle bycatch was reduced using illuminated nets (0.5 individuals/km/day) compared to unlit nets (1.4). Commercially-targeted fish species catch was not affected by LED lighting (illuminated: 10.4 individual fish/km/day, unlit: 10.6). Eleven vessels were equipped with a pair of bottom-set gillnets (56.4 x 2.8 m), one without illumination and the other with green LED lights every 10 m along the float line. Boats set lines for a total of 114 overnight deployments. Pairs of nets were separated by 200 m to avoid lighting the control nets. The catch of sea turtles was recorded on board.

A replicated study in 1992–2015 in pelagic longline fisheries in the Atlantic and North Pacific (3) found that using more light sticks on longlines resulted in a higher chance of catching loggerhead turtles *Caretta caretta* but had no impact on leatherbacks *Dermochelys coriacea* (data reported as statistical model results). Pelagic Observer Program data from (1992–2015) was used to determine the number of turtles caught/1,000 hooks, and variation in the number of light sticks/hook (average of 0.4–0.9 sticks/hook) was used to test its effect on bycatch.

A randomized, controlled, paired study in 2015–2016 in sandy-muddy bottom habitat in the north Adriatic Sea, central Mediterranean Sea (4) found that using UV lights on bottom-set gillnets led to fewer loggerhead turtles *Caretta caretta* being caught. No statistical tests were carried out. No turtles were caught in lit gillnets, compared to 16 individuals in unlit gillnets (1 turtle/1,000 m net length/12 h). Five turtles died after being caught. Catch rates of commercially-targeted fish were similar between lit nets (15 individuals/1,000 m net length/12 h; 17 kg catch/1,000 m net length/12 hours) and unlit nets (14 individuals/1,000 m net length/12 hours soaking time; 17 kg catch/1,000 m net length/12 hours soak time). Data were collected in June–July 2015–2016 during 18 fishing trials. Fishing gear included bottom-set gillnets (average depth of deployment: 54 m) comprising connected netting panels (mesh size: 140 mm, panel length: 100 m, 3 m stretched drop). UV LED lights were positioned 15 m apart along the top line ('floatline') of some of the net panels (70 lights/km). Lit (3 km average net length) and unlit panels (1 km average net length) were randomly distributed along each net. A gap of 150 m was left between lit and unlit panels. Nets deployed from a single fishing vessel (18:00–06:00 h; average soak time: 15 hours). Catch of target, discard and unwanted species was monitored.

(1) Wang J.H., Fisler S. & Swimmer Y. (2010) Developing visual deterrents to reduce sea turtle bycatch in gill net fisheries. *Marine Ecology Progress* Series, 408, 241–250.
(2) Ortiz N., Mangel J.C., Wang J., Alfaro-Shigueto J., Pingo S., Jimenez A. & Godley B.J. (2016) Reducing green turtle bycatch in small-scale fisheries using illuminated gillnets: the cost of saving a sea turtle. *Marine Ecology Progress Series*, 545, 251–259.
(3) Swimmer Y., Gutierrez A., Bigelow K., Barceló C., Schroeder B., Keene K., Shattenkirk K. & Foster D.G. (2017) Sea turtle bycatch mitigation in U.S. longline fisheries. *Frontiers in Marine Science*, 4, 260.
(4) Virgili M., Vasapollo C. & Lucchetti A. (2018) Can ultraviolet illumination reduce sea turtle bycatch in Mediterranean set net fisheries? *Fisheries Research*, 199, 1–7.

6.18. Retain buoys and lines at the sea floor or riverbed when not hauling

- We found no studies that evaluated the effects on reptile populations of retaining buoys and lines at the sea floor or riverbed when not hauling.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Retaining buoys and lines at the sea floor or riverbed when not hauling may reduce the risk of marine and freshwater reptiles becoming entangled in vertical lines within the water column. Buoy lines may be kept coiled on the fishing pot or trap until they are remotely released to the surface by fishers for hauling (e.g. Partan & Ball 2016). Automatic or timed-release systems may also be used.

Partan J. & Ball K. (2016) *Rope-less fishing technology development*. Project 5 Final Report, Consortium for Wildlife Bycatch Reduction.

6.19. Retain offal on fishing vessels instead of discarding overboard

- We found no studies that evaluated the effects on reptile populations of retaining offal on fishing vessels instead of discarding overboard.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Many fishing boats prepare fish onboard, after catching them, in order to maximise the catch that can be stored. The offal (waste) is then normally discarded overboard. Discarding offal overboard during fishing may attract aquatic reptiles and increase the risk of entanglement or capture in gear. Retaining offal on board or disposing of offal at locations and times away from fishing operations may reduce this risk.

6.20. Set gillnets perpendicular to the shore

- We found no studies that evaluated the effects on reptile populations of setting gillnets perpendicular to the shore.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Gillnets hang in the water at various depths and may entangle reptiles. Setting gillnets perpendicular to the shoreline may reduce the chance of entanglement with reptiles (especially sea turtles) that are approaching the beach to nest or returning to the ocean following nesting or hatching.

6.21. Promote knowledge exchange between fishers to improve good practice

- **One study** evaluated the effects on reptile populations of promoting knowledge exchange between fishers to improve good practice. This study was in the USA[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Survival (1 study):** One before-and-after study in the USA[1] found that following the introduction of a tool to help facilitate knowledge exchange and the avoidance of loggerhead turtles, loggerhead turtle bycatch was similar compared to the two years before the tool was introduced.

BEHAVIOUR (0 STUDIES)

OTHER (1 STUDY)

Human behaviour change (1 study): One before-and-after study in the USA[1] found that following the introduction of a tool to help facilitate avoidance of loggerhead turtles, fishers did not spend less time fishing in the areas recommended for avoidance by the tool.

Background

Fishers commonly share information through social relationships, and this may lead to increased fishing success (Turner *et al*. 2014). However, knowledge exchange between fishers could also be used as a tool to help promote sustainable practices, particularly if the central fishers in information-sharing networks can be identified and co-opted to assist managers in the spreading of conservation information (Mbaru & Barnes 2017).

Mbaru E.K. & Barnes M.L. (2017) Key players in conservation diffusion: Using social network analysis to identify critical injection points. *Biological Conservation*, 210, 222–232.

Turner R.A., Polunin N.V.C. & Stead S.M. (2014) Social networks and fishers' behavior: Exploring the links between information flow and fishing success in the Northumberland lobster fishery. *Ecology and Society,* 19, 38.

A before-and-after study in 2005–2007 in pelagic waters north of Hawaii, USA (1) found that a tool ('TurtleWatch') created to facilitate knowledge exchange and the avoidance of loggerhead turtle *Caretta caretta* interactions with a swordfish *Xiphias gladius* shallow-set longline fishery, did not reduce turtle catch, and fishers did not spend less time fishing in areas recommended for avoidance by the tool. Results were not statistically tested. After the tool was deployed, 0–0.03 loggerhead turtles/1000 hooks (12 total turtles) were caught compared to 0.01–0.04 loggerhead turtles/1000 hooks (17 total turtles) in the previous year and 0–0.05 turtles/1000 hooks (9 total turtles) two years earlier. Fishers did not remain south of the fishing boundary line recommended by the tool, instead the whole fishery moved further north than previously and remained north for a longer time than in the two preceding years (see paper for details). 'TurtleWatch' combined historical fishing, environmental and turtle behavioural data to recommend areas to avoid fishing. In January–March 2007, information from the tool was disseminated daily in electronic and paper format to industry professionals and fishers. The fishery also had a legal catch limit of 17 turtles/year, after which fishery closures were imposed. In January–March 2005–2007, line deployments (2005: 520 deployments; 2006: 842; 2007: 797), number of hooks put out (2005: 429,580 hooks; 2006: 670,914; 2007: 689,486), and loggerhead turtle interactions were monitored.

(1) Howell E.A., Kobayashi D.R., Parker D.M., Balazs G.H. & Polovina J.J. (2008) TurtleWatch: A tool to aid in the bycatch reduction of loggerhead turtles *Caretta caretta* in the Hawaii-based pelagic longline fishery. *Endangered Species Research*, 5, 267–278.

Hooks, lines, nets and traps

6.22. Use circle hooks instead of J-hooks

- **Eleven studies** evaluated the effects of using circle hooks instead of J-hooks on reptile populations. Five studies were in the Atlantic[1,5,6,8,9], three were in the Pacific[2,7,10] and one study was in each of the Mediterranean[4], the Atlantic and North Pacific[11] and the western North Atlantic, Azores, Gulf of Mexico and Ecuador[3].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (3 STUDIES)

- **Survival (3 studies):** Two studies (including one replicated, controlled, paired study) off the coast of Hawaii[2] and in the north-east Atlantic Ocean[9] found that survival of loggerhead and leatherback turtles[2] and leatherback and hard-shell sea turtles[9] caught by circle hooks or J-hooks was similar. One review of studies in five pelagic longline fisheries[3] found that fewer sea turtles died when circle hooks were used compared to J-hooks in four of five fisheries.
- **Condition (3 studies):** Two replicated, controlled studies in the Mediterranean Sea[4] and south-western Atlantic Ocean[5] found that fewer immature loggerhead turtles[4] and loggerhead turtles[5] swallowed circle hooks compared to J-hooks. One before-and-after study off the coast of Hawaii[2] found that a lower percentage of loggerhead and leatherback turtles were deeply hooked by circle hooks compared to J-hooks.

BEHAVIOUR (0 STUDIES)

OTHER (11 STUDIES)

- **Unwanted catch (11 studies):** Seven of 10 studies (including six replicated, controlled studies) in the Pacific[2,7,10], Atlantic[1,5,6,8,9], Atlantic and North Pacific[11] and Mediterranean[4] and one review of studies in five pelagic longline fisheries[3] found that circle hooks or circle hooks and tuna hooks[10] caught fewer sea turtles than J-hooks[1,4,5,7,10,11], or that non-offset G-style circle hooks caught fewer leatherback and hard-shell sea turtles that offset J-Hooks[9]. One of these studies[5] also found that circle hooks caught slightly larger loggerhead turtles than J-hooks, and one[9] also found that offset Gt-style circle hooks caught a similar number of leatherback and hard-shell sea turtles compared to offset J-hooks. One study[6] found that circle hooks caught a similar number of leatherback, green and olive ridley turtles compared to J-hooks. One study[2] found that fish-baited circle hooks caught fewer loggerhead and leatherback turtles than squid-baited J-hooks. The review[3] found mixed effects of using circle hooks compared to J-hooks on unwanted catch of sea turtles depending on the fishery. The other study[8] found mixed effects of using circle hooks or J-hooks in combination with squid or fish bait on the number of loggerhead and leatherback turtles that were caught.

Background

Sea turtles are vulnerable to being hooked in the mouth or swallowing hooks on longline fishing hooks when foraging for bait attached to the hooks. They may also be hooked in the body or become entangled in the longlines when swimming in the vicinity of lines that have been set.

Three main types of fishing hooks are typically used in longline fisheries: J-hooks, tuna hooks, and circle hooks. J-hooks are shaped like a 'J', with the hook point parallel to the hook shaft. Circle hooks are more rounded than J-hooks with the hook point turned in so that it is at right-angles to the hook shaft (FAO 2009). Tuna hooks are shaped in between a J-hook and a circle hook.

Using circle hooks may reduce unwanted catch compared to J-hooks because they are wider and so may be less likely to fit into a turtle's mouth. They may also reduce the incidences of turtles swallowing hooks, thereby increasing the chances of turtles surviving after being caught and released (Ryder *et al*. 2006).

This action includes studies that discuss comparisons between different types of circle hook with different types of J-hook. See *Use non-offset hooks*, *Use non-ringed hooks*, and *Use larger hooks* for studies that specifically test these variations in hook design. See also *Modify number of hooks between floats on longlines*.

FAO (2009) *Guidelines to reduce sea turtle mortality in fishing operations*. FAO Fisheries Department, Rome.

Ryder C.E., Conant T.A. & Schroeder B.A. (2006) *Report of the workshop on marine turtle longline post-interaction mortality*. NOAA Technical Memorandum NMFS-OPR-29.

A replicated, controlled study in 2002 in pelagic waters in the north-western Atlantic Ocean (1) found that using 18/0 circle hooks with squid *Illex* spp. or mackerel *Scomber scombrus* bait instead of J-hooks reduced unwanted catch of sea turtles in a tuna and swordfish *Xiphias gladius* longline fishery. Mackerel-baited circle hooks reduced loggerhead *Caretta caretta* catch by 90% (0.04 turtles/1,000 hooks), squid-baited circle hooks by 86% (0.05 turtles/1,000 hooks), and mackerel-baited J-hooks by 71% (0.13 turtles/1,000 hooks) compared to when squid-baited J-hooks were used (0.5 turtles/1,000 hooks). Mackerel-baited circle hooks reduced leatherback turtle *Dermochelys coriacea* catch by 65% (0.15 turtles/1,000 hooks), squid-baited circle hooks by 57% (0.21 turtles/1,000 hooks), mackerel-baited J-hooks by 66% (0.15 turtles/1,000 hooks) compared to squid-baited J-hooks (0.50 turtles/1,000 hooks). Most (55 of 80) loggerheads caught swallowed J-hooks, while few swallowed circle hooks (3 of 11, results were not statistically tested). No leatherback turtles swallowed either hook type. Five hook/bait combinations were trialled: 0° offset 18/0 circle hooks with 150–300 g squid bait; 10° offset 18/0 circle hooks with squid bait; 20°–25° offset 9/0 J-hooks with 200–500 g mackerel bait; 10° offset 18/0 circle hooks with mackerel bait; and 20°–25° offset 9/0 J-hooks with squid bait (standard in the fishery). Thirteen vessels made 489 deployments, fishing a total of 427,382 hooks (71,000 hooks/bait for each of the four new combinations and 142,000 hooks for the standard combination). On-board observers collected catch data.

A before-and-after study in 1994–2006 in pelagic waters off the coast of Hawaii, USA (2) found that fish-baited circle hooks reduced unwanted catch of sea turtles compared to squid-baited J-hooks in a swordfish *Xiphias gladius* longline fishery. Capture rates of leatherback *Dermochelys coriacea* reduced by 83% (0.006 turtles/1,000 hooks) and loggerhead *Caretta caretta* turtles by 90% (0.012 turtles/1,000 hooks) when fish-baited circle hooks were used compared to squid-baited J-hooks (leatherback: 0.03 turtles/1,000 hooks, loggerhead: 0.13 turtles/1,000 hooks). Mortality rates were similar whether circle (35 of 35 turtles survived) or J-hooks (180 of 182 survived) were used. Fewer turtles were deeply hooked when circle hooks were used (leatherback: 0%, hard-shell: 22%) compared to J-hooks (10%, 60%). Target swordfish catch increased by 16% after circle hooks were introduced, but tuna (Scombridae spp.), mahi mahi *Coryphaena* spp., opah *Lampris* spp. and wahoo *Acanthocybium solandri* catch reduced by 34–50% (see paper for details). Catch data from the US National Marine Fisheries Service observer programme were compared from before and after regulations were introduced requiring the use of 10° offset 18/0 circle hooks with fish bait in a pelagic swordfish longline fishery. Prior to the regulations, 9/0 J-hooks with squid bait were used. 'Before' data used was from 1994–2002 (120 observed trips of 1,631 sets with 1,282,748 J hooks deployed) and 'after' data was from 2004–2006 (164 observed trips of 2,631 sets with 2,150,674 hooks deployed).

A review of studies in 2000–2004 in five pelagic longline fisheries in the western North Atlantic, Azores, Gulf of Mexico and Ecuador (3) found that using circle hooks instead of traditional J-hooks reduced overall unwanted catch in three of five fisheries and mortality rates of sea turtles in four of the fisheries. Unwanted catch reduced significantly in two of five fisheries and in one of four years in a third fishery. Sea turtle mortality rates reduced significantly in four of five fisheries. Switching to circle hooks from J-hooks was considered economically viable in three of five fisheries, not viable in a fourth (as target catch was reduced significantly) and the impact was unknown in the fifth. The fisheries were for tuna *Thunnus* spp. and mahi mahi *Coryphaena hippurus*. Experiments comparing use of circle hooks (offset and non-offset of different sizes, see original paper) with traditional J-hooks were carried out in 2000–2004 on longline vessels (1–136 vessels/fishery, 48–489 deployments/fishery with 20,570–578,050 hook deployments/fishery).

A replicated, controlled study in 2005–2007 in pelagic waters in the Mediterranean Sea, Italy and Tunisia (4) found that circle hooks caught fewer immature loggerhead turtles *Caretta caretta* than J-hooks in a shallow-set swordfish *Xiphias gladius* longline fishery. Unwanted catch of immature sea turtles was lower when circle hooks (0.4 individuals/1,000 hooks) were used compared to J-hooks (1.4). Five of 20 turtles swallowed J-hooks, compared to none of six turtles caught with circle hooks (results were not statistically tested). Catch rates of commercially targeted swordfish were similar between hook types (circle: 13 individuals/1,000 hooks, J: 15). Catch rates of 10° offset 16/0 circle hooks (2.7 cm gape width) were compared with traditional 20° offset size 2 J-hooks (2.6 cm gape width). Seven experimental trips were conducted using a single commercial fishing boat, totalling 30 fishing sets in July–October 2005–2007. Circle and J-hooks were alternated along the mainline (30,000 total hooks, 50% of each type).

A replicated, controlled study in 2004–2008 in pelagic waters in the south-western Atlantic Ocean in Brazil (5) found that using circle hooks reduced unwanted catch of sea turtles compared to J-hooks in a longline fishery. Unwanted catch of loggerhead *Caretta caretta* and leatherback *Dermochelys coriacea* were reduced when circle hooks were used (loggerhead: 0.8 turtles/1,000 hooks, leatherback: 0.7) compared to J-hooks (loggerhead: 1.9, leatherback: 1.6). Fewer loggerhead turtles swallowed hooks when circle hooks were used (6%) compared to J-hooks (25%). However, on average, circle hooks caught larger loggerheads (61 cm average carapace length) than J-hooks (58 cm). Catch rates of most target fish species was increased when circle hooks were used, with the exception of swordfish *Xiphius gladius* (see paper for details). Catch rates of 10° offset 18/0 circle hooks (2.8–2.2 cm gape width) were compared to traditional 9/0 0° offset J-hooks (2.9 cm gape width). Twenty-seven trips totalling 229 fishing trips were undertaken. A total of 145,828 baited hooks were tested by alternating hooks along sections of the mainline.

A replicated, paired, controlled study in 2006–2007 in pelagic waters in the western equatorial Atlantic Ocean off the coast of Brazil (6) found that using circle instead of J-hooks in a longline fishery did not reduce unwanted catch of leatherback turtles *Dermochelys coriacea*, green turtles *Chelonia mydas* or olive ridley turtles *Lepidochelys olivacea.* Numbers of sea turtles caught with circle hooks (leatherback: 1.4 turtles/1,000 hooks, green: 1.4, olive ridley: 2.5) was statistically similar to J-hooks (3.1, 1.7, 1.9). Catch rates of commercially-targeted bigeye tuna *Thunnus obesus* increased when circle hooks were used (23 fish/1,000 hooks) compared to J-hooks (17 fish/1,000 hooks). Catch rates of commercially-targeted sailfish *Istiophorus platypterus* reduced when circle hooks were used (0.6 fish/1,000 hooks) compared to J hooks (4.4 fish/1,000 hooks). Catch rates of all other commercially-targeted species were similar between hook types (see paper for details). On six fishing trips, three commercial pelagic longline fishing vessels (24.6–26.9 m long) using similar gear carried out 81 deployments targeting swordfish *Xiphias gladius* and bigeye tuna *Thunnus obseus* (11–15 deployments/trip) in August 2006–January 2007. Circle hooks (size 18/0, 0◦ offset) and traditional J-style hooks (size 9/0, 10◦ offset) were alternated along the mainline (50,170 hooks in total, divided equally between circle and J-hooks). Hooks were baited with squid *Illex* sp. and lit with battery-run light attractants. Lines were deployed overnight.

A replicated, controlled study in 2004–2010 in Ecuadorean, Panamanian and Costa Rican fisheries in the eastern Pacific Ocean (7) found that unwanted catch of sea turtles was reduced when circle hooks were used instead of J-hooks in five artisanal surface longline fisheries. Unwanted catch of sea turtles was reduced when circle hooks were used compared to J-hooks in mahi mahi *Coryphaena hippurus* fisheries in Ecuador (circle: 1.3–1.6 turtles/1,000 hooks, J: 2.0–2.2) and Costa Rica (circle: 2.3, J: 2.9) and in combined tuna *Thunnus albacares*, billfish (Istiophoridae and Xiphiidae) and shark fisheries in Ecuador (circle: 0.6, J: 1.3), Costa Rica (circle: 0.4–1.5, J: 1.3–1.5) and Panama (circle: 0.9, J: 2.0). The effect on target fish species was mixed; in three comparisons circle hooks increased catch, in three they reduced catch and in one there was no difference (see original paper for details). A voluntary program to test use of circle hooks instead of traditional J-hooks began in the Eastern Pacific Ocean in 2004. Unwanted catch of sea turtles was compared between circle hooks (sizes: 14/0–18/0) and traditional J-hooks (J-style or tuna) in mahi mahi fisheries (Ecuador: 2 fisheries; Costa Rica: 1 fishery) and combined tuna, billfish and shark fisheries (Ecuador: 1 fishery; Panama: 1 fishery; Costa Rica: 2 fisheries). Hook sizes, baits, vessels and longline materials varied between fisheries (see original paper). Hook types were placed alternately along the long lines. A total of 3,126 longline deployments were made (328,523 total J-hooks; 401,839 total circle hooks).

A controlled study in 2008–2012 in pelagic waters in the Southern Atlantic (8) found that using circle hooks reduced unwanted catch of loggerhead *Caretta caretta* and leatherback *Dermochelys coriacea* turtles compared to using J-hooks when using squid *Illex* spp. instead of fish *Scomber* spp. bait. When squid was used as bait, the catch of all turtles was lower when using non-offset circle hooks (0.7 turtles/1,000 hooks) and offset circle hooks (0.6 turtles/1,000 hooks) compared to J-hooks (1.7 turtles/1,000 hooks). Total turtle catch was similar when mackerel bait was used (non-offset circle: 0.2 turtles/1,000 hooks; offset circle: 0.2

turtles/1,000 hooks; J-hook: 0.3 turtles/1,000 hooks). This pattern was observed for both leatherback and loggerhead turtles (see original paper for details). Overall turtle survival was higher when offset circle hooks were used (49 of 59, 83% of individuals alive) compared to non-offset circle hooks (38 of 72, 53% of individuals alive) or J-hooks (99 of 155, 64% of individuals alive). This pattern was observed for loggerhead turtles, but leatherback turtle survival was similar between hook types (see original paper for details). Three hook types baited with either squid or mackerel were used alternately on a commercial longline fishing vessel: traditional J-hook (size: 9/0) and two circle hooks (a non-offset and a 10° offset, both sized: 17/0; 148,800 total hooks/type). In total 310 longline deployments (1,440 hooks/deployment; 446,400 total hooks, lines set to 20–50 m depths) were carried out overnight in October 2008–February 2012. One bait type was used in each deployment. Turtle catch was monitored by onboard observers.

A replicated, controlled paired study in 2008–2011 in pelagic waters in the north-east Atlantic Ocean (9) found that changing to non-offset circle hooks from offset J-hooks in a longline swordfish *Xiphias gladius* fishery reduced unwanted catch of sea turtles. Unwanted sea turtle catch was reduced with non-offset G-style circle hooks (leatherback turtles *Dermochelys coriacea*: 0.34–0.50 turtles/1,000 hooks; hard-shell turtles (Cheloniidae spp.): 0.07–0.14), but not offset Gt-style circle hooks (leatherback turtles: 0.73–0.78 turtles/1,000 hooks; hard-shell turtles: 0.07–0.19), compared to traditional offset J-hooks (leatherback turtles: 0.94–0.99 turtles/1,000 hooks; hard-shell turtles: 0.16–0.35). Mortality and hooking location of leatherback turtles was similar between hook types (see paper for details). In August 2008–December 2011, a commercial vessel carried out 202 overnight longline fishing deployments (lines: 55 nm long with 5 branchlines, deployed 20–50 m deep, lit by green lights). Whole squid (*Illex* spp.) or mackerel (*Scomber* spp.) were used as bait (one type of bait/line deployment). Three hook styles: 10° offset J-hooks traditionally used in the fishery; non-offset G-style circle hooks; and 10° offset Gt-style circle hooks were alternated every 70–80 hooks along the line in a randomized start order (254,520 total hooks deployed with 42,420 of each hook/bait combination). Unwanted catch was counted and released.

A replicated study in 2004–2011 in pelagic waters in the Eastern Pacific Ocean (10) found that using circle hooks or tuna hooks instead of traditional J-hooks reduced the likelihood of olive ridley turtles *Lepidochelys olivacea* swallowing hooks in an artisanal surface longline fishery. All results were reported as odds ratios, see original paper for details. Both circle and tuna hooks were less likely, and circle hooks least likely to be swallowed overall, by olive ridley turtles compared to J-hooks. In 2004–2011 incidental sea turtle catch rates of circle hooks (sizes 12/0–18/0), tuna hooks and traditional J-hooks (see original paper for hook specifications) were compared by placing hooks in alternative sequence along longlines (3.5 million total hooks used in 8,996 line deployments). Bait used was classed as squid (*Dosidicus gigas, Illex* sp. and *Loligo* sp.) or fish (*Opisthonema* spp., *Scomber japonicus, Auxis* spp. and *Sardinops sagax*) and only deployments using one type of bait were included in analysis (4,838 of 8,996 line deployments). Information on hooking location and entanglement of sea turtles was recorded (1,823 total olive ridley turtles).

A replicated, before-and-after study in 1992–2015 in pelagic longline fisheries in the Atlantic and North Pacific (11) found that using circle hooks on longlines resulted in less leatherback *Dermochelys coriacea* and loggerhead *Caretta caretta* bycatch compared to when J-hooks were used. The chance of catching turtles on longlines was lower in the Atlantic when circle hooks were used (leatherback: 0–6% chance with fish bait (species not provided), 9% with squid bait (species not provided); loggerhead: 0–5% with fish bait, 11% with squid bait) compared to J-hooks (leatherback: 13% with fish bait, 20% with squid bait; loggerhead: 9% with fish bait, 18% with squid bait). The same was true in the Pacific (leatherback - circle hook: <1% vs. J-hook: 1%; loggerhead: circle hook: 1% with fish, 2% with squid vs. J-hook: 5% with fish, 13% with squid). Following the introduction of regulations on bait and hooks, overall bycatch was reduced in both the Atlantic (leatherback: 40% reduction; loggerhead: 61% reduction) and Pacific (leatherback: 84% reduction; loggerhead 95% reduction). Fisheries were closed in 2001 and re-opened with regulations regarding bait (fish or squid) and hook type (circle or J-hooks) (see paper for details). Pelagic Observer Program data from before (1992–2001) and after (2004–2015) regulations was used to determine the number of turtles caught/1,000 hooks.

(1) Watson J.W., Epperly S.P., Shah A.K. & Foster D.G. (2005) Fishing methods to reduce sea turtle mortality associated with pelagic longlines. *Canadian Journal of Fisheries and Aquatic Sciences*, 62, 965–981.
(2) Gilman E., Kobayashi D., Swenarton T., Brothers N., Dalzell P. & Kinan-Kelly I. (2007) Reducing sea turtle interactions in the Hawaii-based longline swordfish fishery. *Biological Conservation*, 139, 19–28.
(3) Read A.J. (2007) Do circle hooks reduce the mortality of sea turtles in pelagic longlines? A review of recent experiments. *Biological Conservation*, 135, 155–169.
(4) Piovano S., Swimmer Y. & Giacoma C. (2009) Are circle hooks effective in reducing incidental captures of loggerhead sea turtles in a Mediterranean longline fishery? *Aquatic Conservation: Marine and Freshwater Ecosystems*, 19, 779–785.
(5) Sales G., Giffoni B.B., Fiedler F.N., Azevedo V.G., Kotas J.E., Swimmer Y. & Bugoni L. (2010) Circle hook effectiveness for the mitigation of sea turtle bycatch and capture of target species in a Brazilian pelagic longline fishery. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 20, 428–436.
(6) Pacheco J.C., Kerstetter D.W., Hazin F.H., Hazin H., Segundo R.S.S.L., Graves J.E., Carvalho F. & Travassos P.E. (2011) A comparison of circle hook and J hook performance in a western equatorial Atlantic Ocean pelagic longline fishery. *Fisheries Research*, 107, 39–45.
(7) Andraka S., Mug M., Hall M., Pons M., Pacheco L., Parrales M., Rendón L., Parga M.L., Mituhasi T., Segura Á., Ortega D., Villagrán E., Pérez S., Paz C. de, Siu S., Gadea V., Caicedo J., Zapata L.A., Martínez J., Guerrero P., Valqui M. & Vogel N. (2013) Circle hooks: Developing better fishing practices in the artisanal longline fisheries of the Eastern Pacific Ocean. *Biological Conservation*, 160, 214–224.
(8) Santos M.N., Coelho R., Fernandez-Carvalho J. & Amorim S. (2013) Effects of 17/0 circle hooks and bait on sea turtles bycatch in a Southern Atlantic swordfish longline fishery. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 23, 732–744.
(9) Coelho R., Santos M.N., Fernandez-Carvalho J. & Amorim S. (2015) Effects of hook and bait in a tropical northeast Atlantic pelagic longline fishery: Part I-Incidental sea turtle bycatch. *Fisheries Research*, 164, 302–311.
(10) Parga M.L., Pons M., Andraka S., Rendon L., Mituhasi T., Hall M., Pacheco L., Segura A., Osmond M. & Vogel N. (2015) Hooking locations in sea turtles incidentally captured by artisanal longline fisheries in the Eastern Pacific Ocean. *Fisheries Research*, 164, 231–237.
(11) Swimmer Y., Gutierrez A., Bigelow K., Barceló C., Schroeder B., Keene K., Shattenkirk K. & Foster D.G. (2017) Sea turtle bycatch mitigation in U.S. longline fisheries. *Frontiers in Marine Science*, 4, 260.

6.23. Use non-offset hooks

- **Two studies** evaluated the effects of using non-offset hooks on reptile populations. One study was off the Pacific coast of Costa Rica[1] and one was in the north-east Atlantic[2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Survival (1 study):** One replicated, controlled, paired study in the north-east Atlantic Ocean[2] found that mortality of leatherback turtles was similar when caught with non-offset hooks or offset hooks.

BEHAVIOUR (0 STUDIES)

OTHER (2 STUDIES)

- **Unwanted catch (2 studies):** One of two replicated, paired studies (including one controlled study) off the Pacific coast of Costa Rica[1] and in the north-east Atlantic[2] found that non-offset circle hooks caught a similar number of olive ridley and green turtles compared to offset circle hooks in a shallow-set longline fishery[1]. The other study[2] found that non-offset G-style circle hooks caught fewer leatherback and hard-shell turtles compared to offset Gt-style circle hooks or offset J-hooks in a longline swordfish fishery.

Background

The hook points on non-offset hooks lie along the same plane as the hook shaft, whereas the points on offset hooks are bent sideways and out of alignment with the hook shaft (usually within 25°). As a result, offset hooks may be more likely to catch turtles in the mouth or on the body than non-off set hooks.

Studies in this action specifically test whether non-offset hooks are more, or less likely to catch sea turtles than offset hooks. For studies that compare using different types of circle hooks to different types of J-hooks, see: *Use circle hooks instead of J-hooks*.

See *Use non-ringed hooks* and *Use larger hooks* for studies that specifically test the effect of using these variations in hook design. See also See also *Modify number of hooks between floats on longlines*.

A replicated, paired study in 2004–2006 in pelagic waters in the Gulf of Papagayo, Costa Rica (1) found that using non-offset circle hooks did not reduce unwanted catch rates of sea turtles compared to offset circle hooks in a shallow-set longline fishery. Catch rates of olive ridley *Lepidochelys olivacea* and green *Chelonia mydas* sea turtles were similar when non-offset circle hooks were used (olive ridley: 19.1 turtles/1,000 hooks, green: 0.3) compared to offset circle hooks (18.9, 0.4). Catch rates of commercially-targeted dolphinfish *Coryphaena* spp. were similar between hook types (non-offset: 53.1 fish/1,000 hooks, offset: 51.3). Circle hooks (size: 14/0) with and without a 10° offset point relative to the shaft of the hook were tested during six fishing trips with 42 shallow-set longline deployments (33,876 total hooks, 800 hooks/day) in November–March, 2004–2006. Hook types were alternated along each longline. Humboldt squid *Dosidicus gigas* was used as bait (approximately 50 x 50 x 250 mm sized pieces). Lines were deployed in the mornings and hauled in after 12 hours. Sea turtle catch was monitored by onboard observers.

A replicated, controlled, paired study in 2008–2011 in pelagic waters in the north-east Atlantic Ocean (2) found that using non-offset circle hooks instead of offset J-hooks in a longline swordfish *Xiphias gladius* fishery reduced unwanted catch of sea turtles. Unwanted sea turtle catch was reduced with non-offset G-style circle hooks (leatherback turtles *Dermochelys coriacea*: 0.3–0.5 turtles/1,000 hooks; hard-shell sea turtles [Cheloniidae spp.]: 0.1), but not offset Gt-style circle hooks (leatherback turtles: 0.7–0.8 turtles/1,000 hooks; hard-shell turtles: 0.1–0.2), compared to traditional offset J-hooks (leatherback turtles: 0.9–1.0 turtles/1,000 hooks; hard-shell turtles: 0.2–0.4). Mortality and hooking location of leatherback turtles was similar between hook types (see paper for details). In August 2008–December 2011, a commercial vessel carried out 202 overnight longline fishing deployments (lines: 55 nm long with 5 branchlines, deployed 20–50 m deep, lit by green lights). Whole squid (*Illex* spp.) or mackerel (*Scomber* spp.) were used as bait (one type of bait/line deployment). Three hook styles: 10° offset J-hooks traditionally used in the fishery; non-offset G-style circle hooks; and 10° offset Gt-style circle hooks were alternated every 70–80 hooks along the line in a randomized start order (254,520 total hooks deployed with 42,420 of each hook/bait combination). Unwanted catch was counted and released.

(1) Swimmer Y., Arauz R., Wang J., Suter J., Musyl M., Bolaños A. & López A. (2010) Comparing the effects of offset and non-offset circle hooks on catch rates of fish and sea turtles in a shallow longline fishery. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 20, 445–451.

(2) Coelho R., Santos M.N., Fernandez-Carvalho J. & Amorim S. (2015) Effects of hook and bait in a tropical northeast Atlantic pelagic longline fishery: Part I-Incidental sea turtle bycatch. *Fisheries Research*, 164, 302–311.

6.24. Use non-ringed hooks

- **One study** evaluated the effects of using non-ringed hooks on reptile populations. This study was in the Mediterranean[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (0 STUDIES)

OTHER (1 STUDY)

- **Unwanted catch (1 study):** One replicated, paired study in the Mediterranean[1] found that when non-ringed circle hooks were used in a swordfish longline fishery fewer loggerhead turtles were caught compared to when ringed hooks were used.

Background

Ringed hooks are made of two parts: the hook itself which is connected to a separate ring that attaches to the line. Non-ringed hooks are in one piece that attaches directly to the line. Ringed hooks tend to be more mobile when underwater and this added mobility may increase the likelihood of sea turtles being caught.

Studies in this action specifically test whether non-ringed hooks are more or less likely to catch sea turtles than ringed hooks. For studies that compare using different types of circle hooks to different types of J-hooks, see: *Use circle hooks instead of J-hooks*.

See also *Use non-offset hooks* and *Use larger hooks* for studies that specifically test these variations in hook design. See also *Modify number of hooks between floats on longlines*.

A replicated, paired study in 2009–2013 in pelagic waters in the Strait of Sicily and South Tyrrhenian Sea, central Mediterranean Sea (1) found that fewer loggerhead turtles *Caretta caretta* were incidentally caught on non-ringed circle hooks than ringed circle hooks in a longline fishery targeting swordfish *Xiphias gladius*. No loggerhead turtles were caught on non-ringed hooks, compared to six turtles caught on ringed hooks (statistical analyses not carried out due to small sample size). Catch rates of target swordfish were lower on non-ringed hooks (7 fish/1,000 hooks) compared to ringed hooks (9 fish/1,000 hooks). Ringed and non-ringed circle hooks (size: 16/0) with a 10° offset were alternately set along a mainline in an even ratio from six longline vessels (600–1,100 hooks/vessel). Data were collected during 65 longline deployments (using 25,400 of each hook type) in July-September in 2009–2010 and 2012–2013.

(1) Piovano S. & Swimmer Y. (2017) Effects of a hook ring on catch and bycatch in a Mediterranean swordfish longline fishery: small addition with potentially large consequences. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 27, 372–380.

6.25. Use larger hooks

- **Two studies** evaluated the effects of using larger hooks on reptile populations. One study was in the USA[1] and one was in the Eastern Pacific Ocean[2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One study in the USA[1] of captive loggerhead turtles found that turtles were less likely to attempt to swallow larger circle hooks than smaller ones.

OTHER (1 STUDY)

- **Unwanted catch (1 study):** One replicated study in the Eastern Pacific Ocean[2] found that olive ridley turtles were less likely to be caught by swallowing larger hooks than smaller ones.

Background

Larger hooks may be more difficult for sea turtles to swallow and so using larger hooks may reduce numbers of hooks being swallowed, which may reduce unwanted catch rates and increase the chances of turtles surviving after being caught and released.

Studies in this action specifically test whether larger hooks are more or less likely to catch sea turtles than smaller hooks. For studies that compare using different types of circle hooks to J-hooks, see *Use circle hooks instead of J-hooks*.

See also: *Use non-ringed hooks* and *Use non-offset hooks* for studies that specifically test the effect of using these variations in hook design. See also *Modify number of hooks between floats on longlines*.

A study in 2004–2005 in laboratory conditions in Texas, USA (1) found that loggerhead turtles *Caretta caretta* were more likely to attempt to swallow smaller circle hooks than larger circle hooks. All results were presented as model outputs, see original paper. The odds that turtles would attempt to swallow the smallest hook was 97 times higher than the odds that they would attempt to swallow the largest hook size, regardless of size of turtle. Larger turtles were more likely to attempt to swallow larger hook sizes. Turtle responses to individual baited hooks suspended in their tanks were video recorded (each hook presentation = 1 trial). Sixty 45 cm long captive-reared turtles participated in trials, of which 30 turtles participated again when they reached 55 cm and 65 cm long. Trials were carried out in April and October 2004, and May 2005. Modified circle hooks of different sizes were trialled: 14/0, 16/0, 18/0 and 20/0 (20 turtles/hook size, 20/0 was not tested with 45 cm turtles). Hooks were baited with whole squid *Illex illecebrosus* or sardines *Sardinella aurita* and either single-baited or 'thread'-baited (see paper for details).

A replicated study in 2004–2011 in pelagic waters in the Eastern Pacific Ocean (2) found that using larger hooks reduced the likelihood of olive ridley turtles *Lepidochelys olivacea* swallowing hooks in an artisanal surface longline fishery. All results were reported as odds ratios, see original paper for details. Overall, larger hook sizes were less likely to be swallowed than smaller hook sizes. Using fish bait in combination with larger circle hooks lead to the largest proportion of external hookings (which are preferable to internal hookings). In 2004–2011 incidental sea turtle catch rates of circle hooks (sizes 12/0–18/0), tuna hooks and traditional J-hooks (see original paper for hook specifications) were compared by placing hooks in alternative sequence along longlines (3.5 million total hooks used in 8,996 line deployments). Bait used was classed as squid (*Dosidicus gigas, Illex* sp. and *Loligo* sp.) or fish (*Opisthonema* spp., *Scomber japonicus, Auxis* spp. and *Sardinops sagax*) and only deployments using one type of bait were included in analysis (4,838 of 8,996 line deployments). Information on hooking location and entanglement of sea turtles was recorded (1,823 total olive ridley turtles).

(1) Stokes L.W., Hataway D., Epperly S.P., Shah A.K., Bergmann C.E., Watson J.W. & Higgins B.M. (2011) Hook ingestion rates in loggerhead sea turtles *Caretta caretta* as a function of animal size, hook size, and bait. *Endangered Species Research*, 14, 1–11.

(2) Parga M.L., Pons M., Andraka S., Rendon L., Mituhasi T., Hall M., Pacheco L., Segura A., Osmond M. & Vogel N. (2015) Hooking locations in sea turtles incidentally captured by artisanal longline fisheries in the Eastern Pacific Ocean. *Fisheries Research*, 164, 231–237.

6.26. Modify number of hooks between floats on longlines

- **One study** evaluated the effects of modifying the number of hooks between floats on longlines on reptile populations. This study was in the Atlantic and North Pacific[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (0 STUDIES)

OTHER (1 STUDY)

- **Unwanted catch (1 study):** One replicated study in the Atlantic and North Pacific[1] found that having fewer hooks between floats did not reduce turtle by-catch in the Pacific but had mixed effects in the Atlantic depending on the species.

Background

Using more floats on a longline per hook alters the profile of the longline in the water, which may reduce the opportunities for sea turtles swimming below the surface to become entangled in the line or caught on the hooks.

For studies looking at the effects of different hook designs, see *Use circle hooks instead of J-hooks*, *Use non-offset hooks, Use non-ringed hooks* and *Use larger hooks.*

A replicated study in 1992–2015 in pelagic longline fisheries in the Atlantic and North Pacific (1) found that using fewer hooks between floats on a longline did not reduce turtle by-catch in the Pacific but had mixed effects in the Atlantic depending on the species. All data presented as statistical model results. In the Pacific, by-catch of leatherback *Dermochelys coriacea* and loggerhead *Caretta caretta* turtles was not affected by the number of hooks between floats. In the Atlantic, the chance of catching leatherback turtles was lower with fewer hooks between floats, whereas loggerheads were less likely to be caught when there were fewer (<3 hooks) or more (>5 hooks) hooks between floats (see paper for details). Pelagic Observer Program data from (1992–2015) was used to determine the number of turtles caught/1,000 hooks, and variation in the number of hooks between floats (majority were 3–5 or 4–5 hooks/float) was used to test its effect on bycatch.

(1) Swimmer Y., Gutierrez A., Bigelow K., Barceló C., Schroeder B., Keene K., Shattenkirk K. & Foster D.G. (2017) Sea turtle bycatch mitigation in U.S. longline fisheries. *Frontiers in Marine Science*, 4, 260.

6.27. Use catch and hook protection devices

- We found no studies that evaluated the effects of using catch and hook protection devices on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Catch and hook protection devices may be used to cover caught fish and hooks during hauling to reduce the chance of entanglement with aquatic reptiles. This may include 'net sleeves' which cover caught fish and hooks with the downward pressure of hauling, or triggered devices (e.g. chains, cages, cones etc.) that automatically release when a fish is hooked. This may prevent reptile injury or death due to hooking.

6.28. Install exclusion devices on fishing gear

Background

Aquatic reptiles may become trapped or entangled in fishing nets (e.g. trawl nets, hoop nets, fyke nets) and traps. Exclusion devices, such as grids, mesh, funnels, rings, or rectangular inserts, can be installed in an attempt to reduce the number of non-target animals that are caught.

Due to the number of studies found, this action has been split by species group.

For studies that discuss the effect of using devices that allow reptiles to escape from fishing gear, see *Install escape devices on fishing gear*, and for studies on the effect of using a combination of exclusion and escape devices see *Install exclusion and escape devices on fishing gear.*

Sea turtles

- **Three studies** evaluated the effects of installing exclusion devices on fishing gear on sea turtle populations. One study was in the Gulf of Mexico[1] (USA), one was in the Mid-Atlantic[2] (USA) and one was off the coast of Western Australia[3].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Survival (1 study):** One replicated, before-and-after study in the Gulf of Mexico[1] found that when exclusion grids with escape holes were used in a shrimp trawl fishery there were fewer lethal strandings of loggerhead turtles compared to when grids were not used.

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One controlled study in the Mid-Atlantic[2] found that when exclusion devices were used on scallop dredges there were fewer interactions with sea turtles than when no devices were used.

OTHER (1 STUDY)

- **Unwanted catch (1 study):** One replicated study off the coast of Western Australia[3] found that exclusion grids with escape hatches prevented sea turtles entering trawl nets.

A replicated, before-and-after study in 1980–2000 on beaches in the Gulf of Mexico, Texas, USA (1) found that mandating use of exclusionary grids with escape holes ('turtle excluder device') in a shrimp trawl fishery reduced lethal strandings of loggerhead turtles *Caretta caretta*. Lethal strandings of loggerhead turtles reduced by 7% after turtle excluder device use was mandated in the fishery compared to beforehand (results reported as model outputs). There was not enough data to assess the effect on Kemp's ridley *Lepidochelys kempii* turtles. Data from the Sea Turtle Stranding and Salvage Network was used to analyse changes in the size and number of stranded turtles before excluder devices were mandated in the shrimp trawl fishery (1986–1990) and afterwards (1995–1999).

A controlled study in 2001–2008 of a pelagic area in the Mid-Atlantic, USA (2) found that scallop dredges with chain mats had lower interaction rates with sea turtles than dredges without chain mats. Overall, the interaction rate of dredges with chain mats and sea turtles was estimated to be 86% lower than that of dredges without chain mats (data reported as statistical model results). The author reported a small number of entanglements with dredges with or without chain mats (see original paper). Turtles observed were loggerhead turtles *Caretta caretta* (47), Kemp's ridley turtles *Lepidochelys kempii* (1) or unidentified species (16). Commercial vessels harvested sea scallops *Placopecten magellanicus* using dredges with and without chain mats attached (fishing effort for each not reported). Chain mats (vertical and horizontal chains hung on the dredge bag) became mandatory from September 2006 in part of the fishing area during May–November each year. Observers onboard the fishing vessels recorded turtles interacting with the dredge gear during a total of 125,658 h (approximately 3% of all commercial fishing trips) in 2001–2008.

A replicated study in 2012 in demersal waters off the coast of Western Australia (3) found that exclusion grids with escape hatches ('bycatch reduction device') prevented sea turtles from entering the codend of trawl nets in a tropical teleost fishery. All 11 sea turtles that entered trawl nets modified with an exclusion grid and escape hatch were expelled (downward-facing grid with square mesh net: 6 turtles; upward-facing grid in diamond mesh: 5 turtles) and 9 of 11 turtles exited in <2.5 minutes. Loss of commercially-targeted teleost species from all trawls was 1.2–1.4% of catch. In June–December 2012, the catch (target and unwanted) from three commercial trawl vessels was monitored using in-net and onboard cameras during daylight. Vessels were fitted with either: upward-facing grid and escape hatch with diamond-mesh net (372 trawl hours on 2 vessels), downward-facing grid and escape hatch with diamond-mesh net (559 trawl hours on 2 vessels), or downward-facing grid and escape hatch with square mesh net (389 trawl hours on 1 vessel; see original paper for all specifications). Use of bycatch reduction grids with escape hatches had been mandatory in this fishery since 2006.

(1) Lewison R.L., Crowder L.B. & Shaver D.J. (2003) The impact of turtle excluder devices and fisheries closures on loggerhead and Kemp's ridley strandings in the Western Gulf of Mexico. *Conservation Biology*, 17, 1089–1097.

(2) Murray K.T. (2011) Interactions between sea turtles and dredge gear in the U.S. sea scallop (*Placopecten magellanicus*) fishery, 2001–2008. *Fisheries Research*, 107, 137–146.

(3) Wakefield C.B., Santana-Garcon J., Dorman S.R., Blight S., Denham A., Wakeford J., Molony B.W. & Newman S.J. (2017) Performance of bycatch reduction devices varies for chondrichthyan, reptile, and cetacean mitigation in demersal fish trawls: assimilating subsurface interactions and unaccounted mortality. *ICES Journal of Marine Science*, 74, 343–358.

Tortoises, terrapins, side-necked & softshell turtles

- **Thirteen studies** evaluated the effects of installing exclusion devices on fishing gear on tortoise, terrapin, side-necked & softshell turtle populations. Ten studies were in the USA[1,3-6,9,10a,10b,10c,11], two were in Canada[7,8] and one was in Australia[2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Survival (1 study):** One replicated, randomized, paired, controlled study in the USA[3] found that fewer turtles died in hoop nets with an exclusion device than in unmodified traps.

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One randomized, controlled trial in the USA[9] found mixed effects of crab pot exclusion devices on use of pots by diamondback terrapins depending on the device design.

OTHER (13 STUDIES)

- **Unwanted catch (13 studies):** Eight of 13 controlled studies (including seven replicated, paired studies) in the USA[1,3-6,9,10a,10b,10c,11], Australia[2] and Canada[7,8] found that crab pots[1,5,6,9], fyke nets[4,8], hoop nets[3] and eel traps[2] with exclusion devices caught fewer turtles[3,4,8], diamond back terrapins[1,5,6,9] and short-necked turtles[2] than unmodified gear. Two studies[2,5] also found that modified gear caught smaller short-necked turtles[2] and diamondback terrapins[5] than unmodified gear. Three studies[7,10a,11] found mixed effects of exclusion devices on unwanted catch of turtles[7] and diamondback terrapins[10a,11] depending on the device design. The other two studies[10b,10c] found that that crab pots with wire exclusion devices[10b] or magnetized exclusion devices[10c] caught a similar number of diamondback terrapins compared to unmodified pots. One study[10b] also found that crab pots with wire exclusion devices caught larger diamondback terrapins than pots with plastic exclusion devices.

A replicated, controlled study in 1996–1997 in an estuarine river in Maryland, USA (1) found that after fitting rectangular exclusion devices ('bycatch reduction device') to crab pots, unwanted catch of diamondback terrapins *Malaclemys terrapin* tended to be lower. In 1996, no terrapins were caught in crab pots modified with a 4 × 10 cm exclusion device, compared to 21 terrapins in unmodified pots. In 1997, fourteen terrapins were caught in pots with a 4.5 × 12 cm exclusion device and 56 in pots with a 5 × 10 cm exclusion device, compared to 105 in unmodified pots (results were not statistically tested). Blue crab *Callinectes sapidus* catch was 2 crabs/pot/day lower when 4 x 10 cm devices were used compared to unmodified pots. Neither the 4.5 x 12 cm nor the 5 x 10 cm excluder device affected crab catch. Three sizes of 11-gauge galvanised wire exclusion devices were tested on modified and unmodified crab pots (standard: 60 cm square, tall: 60 x 60 x 180 cm). In 1996, fourteen unmodified and 14 pots modified with a 4 × 10 cm exclusion device were used (50 days total fishing). In 1997, ten unmodified and 20 pots modified with either 4.5 x 12 cm or 5 x 10 cm exclusion devices were used (10 pots/excluder type, 42 total fishing days). Traps were checked and baited daily.

A replicated, controlled study (years not provided) in two upper tidal creeks in New South Wales, Australia (2) found that using 100 mm exclusion rings on eel *Anguilla* spp. traps reduced unwanted catch of short-necked turtles *Emydura macquarii*. Fewer turtles were caught in traps modified with exclusion rings (8 individuals) compared to unmodified traps (54). Most turtles caught in modified traps were smaller than those caught in unmodified traps (see paper for details). Commercially-targeted eel catch (numbers and size of eels) was similar between modified (49 individuals caught overnight, 21,005 g total catch weight) and unmodified traps (25 individuals, 8,535 g). Standard commercial eel traps (50 cm wide x 40 cm high x 90 cm long mesh traps) had 100 mm PVC rings placed in the entrance funnel. In one site three traps with exclusion rings and three unmodified traps were fished overnight and in a second site, four traps with rings and four unmodified traps were fished for 5 h during the day, cleared, and then fished overnight (12 h). Traps were baited with frozen pilchards *Sardinops neopilchardus*.

A replicated, randomized, controlled, paired study in 2006 along a river in Missouri, USA (3) found that hoop nets modified with an excluder device caught fewer turtles and fewer target fish than unmodified hoop nets in a catfish fishery. Modified hoop nets caught fewer turtles (18 turtles caught in 11 of 50 nets) than unmodified nets (166 turtles caught in 33 of 50 nets). Ten of 18 turtles (56%) died in modified nets compared to 101 of 166 turtles (61%) in unmodified nets (results were not statistically tested). Fewer target catfish species were caught in modified nets (15 individuals) compared to unmodified nets (70 individuals). Unmodified hoop nets (six hoops, 90 cm maximum hoop with 38 mm mesh, 3.7 m long) and modified hoop nets (addition of a tight mesh covering the net entrance to reduce the entrance to 30 cm diameter) were deployed in pairs along four river stretches in May–July 2006 (50 nets/type) using a randomized block design. Nets were set for 48 hours at a time over nine weeks. The catch of turtle and commercially targeted catfish species was recorded.

A replicated, paired, controlled study in 2005–2006 in 34 sites in three river systems in Missouri, USA (4) found that modifying fyke nets (a 'bycatch reduction device') by tying ropes across the entrance resulted in fewer turtles being captured compared to when nets were not modified. Fewer turtles were caught in modified nets (331) compared to unmodified nets (1,355). The average number of turtles caught/night was lower for three of nine species in modified (0–0.2 turtles/night) compared to unmodified nets (0.6–1.4 turtles/night) and similar in modified and unmodified nets for the remaining six species (see paper for details). There was no significant difference in the number of fish caught (modified: 478; unmodified: 415), the number of fish species caught (modified: 23; unmodified: 29), or average catch/night (modified: 0–5 fish/night; unmodified: 0–3 fish/night) in modified compared to unmodified nets. The fyke net was modified by tying four braided ropes (3 mm) vertically (38 mm apart) and three horizontally across the entrance. In 2005–2006, pairs of modified and unmodified nets were deployed ≥100 m apart at 34 sites, including rivers, side channels, backwaters and floodplains. Nets were deployed for 24 hours at each site, and all turtles and fish were counted, identified to species and released.

A replicated, paired, controlled study in 2008 in tidal creeks in Virginia, USA (5) found that using a plastic rectangular device (a 'bycatch reduction device') to reduce the size of entry holes to crab pots reduced the unwanted catch of diamondback terrapins *Malaclemys terrapin* in a blue crab *Callinectes sapidus* fishery. Crab pots with devices caught fewer terrapins (0.01 terrapins/pot/day, 2 individuals) compared to pots without devices (0.20 terrapins/pot/day, 46 individuals). Terrapins caught in traps with devices were smaller on average (5.1 cm shell depth) than terrapins caught in traps without devices (4.3 cm shell depth). Commercially-targeted blue crabs caught in pots with devices had 1.5–2.0 mm wider shells than crabs caught in pots without devices. Catch rates and weight of commercially-targeted crabs were similar between pots with and without devices (see original paper for details). Devices were 4.5 x 12 cm plastic rectangles that were fitted on each of the four entrances of a recreational-style crab pot with chimney (see original paper for details). Crab pots were deployed in shallow-water in 10 pairs in two creeks in summer 2008 (one with and one without devices fitted). Traps were baited once a week for four weeks and checked after 48 h. Terrapin catch was only monitored on one creek.

A replicated, paired, controlled study in 2000–2004 in coastal waters in North Carolina, USA (6) found that after rectangular devices were placed at crab pot entrances (a 'bycatch reduction device'), the unwanted catch of diamondback terrapins *Malaclemys terrapin* tended to be lower in a commercial blue crab *Callinectes sapidus* fishery. No terrapins were caught during hard-shell crab fishing and five terrapins were caught during peeler crab fishing in pots modified with excluders (peeler crab 4.3 cm excluder: 0 individuals, 5.0 cm excluder: 2, vertical ties: 3). Hard shell crab catch was lower in pots with smaller excluders (4.0 cm excluder: 1,002 individuals, 4.5 cm excluder: 459) compared to unmodified pots (625–1,270), but similar when pots with the largest excluder were used (365 individuals) compared to unmodified pots (386). Peeler crab catch was similar in modified pots (372–376 individuals) compared to unmodified pots (374). In May–June 2000–2001 and September–November 2000, hard crab were fished for using pots (60 x 60 x 60 cm) in 21 pairs (with and without excluder devices). Three rectangular excluder devices were tested/season: 16 x 4 cm, 16 x 4.5 cm, and 16 x 5 cm (75 fishing days, 3,150 crab pot days). In April–May 2004, peeler crabs were fished in blocks of four pots with either unmodified, or one of three excluders: 16 x 4.3 cm rectangle, 15.2 x 5.1 cm rectangle, or two vertical wire ties/entrance set 7.8 cm apart (19 fishing days, 1,672 total crab pot days).

A replicated, randomized, paired, controlled study in 2010 in a shallow freshwater lake in Ontario, Canada (7, same experimental location as 8) found that adding exclusion bars to fyke nets (a 'bycatch reduction device') reduced turtle bycatch but that adding exclusion rectangles did not. Fyke nets modified with exclusion bars captured fewer turtles (0.03 turtles/hour) compared to unmodified nets (0.1 turtles/hour). In separate trials, nets modified with an exclusion rectangle captured similar numbers of turtles (0.02 turtles/hour) compared to unmodified nets (0.04 turtles/hour). Catch rates of target fish species were similar in exclusion bar nets (2.9 fish/hour), exclusion rectangle nets (2.6 fish/hour) and unmodified nets (2.9 fish/hour). Standard commercial hooped fyke nets (see original paper for details) were set in a shallow freshwater lake (788 ha) in April–June and September–October 2010 in pairs of modified and unmodified nets. Nets were either modified with exclusion bars made of wooden dowels (8 x 1.3 cm spaced 8 cm apart; set at 30 sites in April–June) or an exclusion rectangle made by attaching a hose clamp at the first funnel of the fyke net (18 x 7.5 cm rectangle; set at 15 sites in September–October). Tandem modified and unmodified nets were set fully submerged within 15 m of each other for 8–48 hours at a time.

A replicated, paired, controlled study in 2011–2012 in a shallow freshwater lake in Ontario, Canada (8, same experimental location as 7) found that modifying fyke nets with a rectangular excluder device ('bycatch reduction device') reduced unwanted catch of turtles in a freshwater fishery. In a first-year smaller scale trial, nets modified with exclusion devices caught statistically similar numbers of turtles (3–4 turtles) compared to unmodified nets (11). The catch of target and non-target fish was also statistically similar between modified (109–144 individuals) and unmodified nets (224). However, in the second year larger scale trial, unwanted catch of turtles was lower in modified nets (0.03 turtles/trapping effort) compared to unmodified nets (0.13). Target species catch was also lower in modified nets (0.64 individuals/trapping effort) compared to unmodified nets (0.95). In September 2011, two fyke nets connected by an entrance net (7 hoops/net, 0.91 m diameter) were deployed in a shallow lake (2.8 m average depth, 780 ha total area) in threes: unmodified net, modified with a 22.5 x 5 cm copper rectangle, or modified with 5 cm spaced vertically-oriented bars across the mouth of the net (nine groups of nets in one site). In April–June 2012, the set up was repeated twice at 11 sites, but did not include nets with the barred exclusion device. All target and non-target catch was identified, counted and released.

A randomized, controlled study (years not provided) in a brackish water experimental enclosure in South Carolina, USA (9) found that using vertically-oriented rectangular devices to limit the size of entry holes on crab pots (a 'bycatch reduction device') reduced the number of entries, increased the time taken to enter and reduced the proportion of successful entry attempts by diamondback terrapins *Malaclemys terrapin* to crab pots. A vertically-oriented device reduced the number of entries into the pot (2 entries/terrapin) compared to horizontally-oriented devices and no device, which produced similar results (horizontal: 5; no device: 6 entries/terrapin). Vertically-oriented devices increased the average time taken to enter a pot (58 seconds before entry) compared to no device, whereas time to enter horizontally-oriented devices was similar to no device (horizontal: 32; no device: 19 seconds before entry). The proportion of terrapins that entered a pot after investigating it was reduced when a vertically-oriented device was used (0.1 terrapins entered/investigation), compared to a horizontally-oriented device (0.2 terrapins entered/investigation). Both types of device reduced the rate of terrapins entering pots compared to no device (0.3 terrapins entered/investigation). In total, 38 wild terrapins were caught to take part in the study, all of a size where they could enter a crab pot when an opening limiting device was present. Each terrapin participated in three randomly ordered trials: vertically-oriented device fitted to entry holes, horizontally-oriented device fitted to entry holes and no device. Devices were 5.1 x 15.2 cm. Crab pots with chimneys baited with mackerel were used. Terrapins were monitored by webcam in 3 h videos (27 h total footage, 3 h/treatment/study group).

A replicated, randomized, controlled study in 2014–2015 in three brackish tidal creeks and a captive setting in Virginia, USA (10a) found that modifying crab pots with red-painted rectangular funnels to reduce the size of entry holes (a 'bycatch reduction device') reduced the unwanted catch of diamondback terrapins

Malaclemys terrapin compared to unmodified pots in a blue crab *Callinectes sapidus* fishery. Crab pots with red-painted funnels set in three tidal creeks caught fewer terrapins (10 individuals) than unmodified pots (58 individuals; no statistical tests were carried out). Trials in both a captive setting and in two tidal creeks found that red-painted funnels also reduced unwanted terrapin catch compared to unmodified pots and that orange, green and blue-painted funnels caught a similar number of terrapins to unmodified pots (see original paper for details). For crab pots set in three creeks, commercially-targeted legal-size blue crab catch was similar in pots with red-painted funnels (622 individuals) compared to unmodified pots (630 individuals). In a captive setting, crabs stayed in pots with funnels for longer (4 h, 45 individuals) compared to unmodified pots (1 h, 76 individuals). Red plastic rectangular funnels (5.1 x 15.2 cm) were fitted horizontally to each of the four entry points on 15 commercial-style crab pots with chimneys. The 15 modified pots were deployed paired with 15 unmodified pots in June–July 2015 in three creeks (587 trap nights, 3–6 pairs of pots/creek). All pots were baited. In separate trails in two creeks (June–July 2014) and in a captive setting (June 2015), orange, black, blue, green-painted and magnetized funnels were also tested (see original paper for details). The captive setting was a seawater tank and crabs and terrapins were monitored by video.

A replicated, controlled study in 2014 in two brackish tidal creeks in Virginia, USA (10b) found that modified crab pots with wire rectangular funnels to reduce the size of entry holes (a 'bycatch reduction device') caught similar numbers of diamondback terrapins *Malaclemys terrapin* compared to unmodified pots and larger-sized terrapins compared to plastic rectangular funnels in a blue crab *Callinectes sapidus* fishery. Crab pots with wire funnels caught similar numbers of terrapins (22 individuals) compared to unmodified pots (20 individuals; no statistical tests were carried out). Terrapins caught in pots with wire funnels were larger (5.5 cm shell height) compared to terrapins caught in pots with plastic funnels (4.8 cm shell height). Commercially-targeted legal-size blue crab catch was similar in pots with wire funnels compared to unmodified pots (see original paper for details). Copper wire or plastic rectangular funnels (5.1 x 15.2 cm) were fitted horizontally to each of the four entry points on commercial-style crab pots with chimneys (five with copper wire funnels, 15 with plastic funnels, five with plastic magnetized funnels, and five unmodified pots). In total five groups of modified and unmodified pots were deployed in June–July 2014 at least 50 m apart in two locations (327 trap days). All pots were baited.

A replicated, randomized, controlled study in 2014–2015 in brackish tidal creeks and a captive setting in Virginia, USA (10c) found that modified crab pots with magnetized rectangular funnels to reduce the size of entry holes (a 'bycatch reduction device') caught similar numbers of diamondback terrapins *Malaclemys terrapin* compared to unmodified pots in a blue crab *Callinectes sapidus* fishery. Crab pots set in two creeks with magnetized funnels caught similar numbers of terrapins (15 individuals) compared to unmodified pots (20 individuals; no statistical tests were carried out). Trials in a captive setting found similar results (see original paper for details). Commercially-targeted legal-size blue crab catch was similar in pots with magnetized funnels compared to unmodified pots (see original paper for details). Rectangular funnels (5.1 x 15.2 cm) were fitted horizontally to each of the four entry points on commercial-style crab pots with chimneys (four with magnetized funnels, 5 unmodified pots). Modified and unmodified pots were deployed in June–July 2014 at least 50 m apart in two creeks (327 trap days, 1–3 pots of each type/creek). All pots were baited. Separate trials in a captive setting were carried out in June 2015 and took place in a seawater tank and crabs and terrapins were monitored by video.

A replicated, controlled, paired study in 2012–2013 in five estuarine sites in North Carolina, USA (11) found that crab pots fitted with one of two different sized wire rectangles ('bycatch reduction devices') limiting the size of the pot opening reduced the numbers of diamondback terrapins *Malaclemys terrapin* caught, compared to unmodified pots. No statistical analyses were carried out due to a small sample size. Pots modified to have small-sized openings and large-sized openings caught zero and one terrapin respectively, compared to 13 terrapins in unmodified pots. None of the terrapins caught were small enough to enter pots with the small-sized opening. One terrapin died in an unmodified pot. Standard commercial crab pots (61 × 61 × 61 cm) were modified with galvanized wire to create large-sized openings (5.1 × 15.2 cm, 10 pots), small-sized openings (3.8 × 15.2 cm, 10 pots) or were unmodified (20 pots). Pots were deployed June–July in pairs (one modified pot and one unmodified pot in 4 sites) in 2012 or in triplicate (1 of each size of modified pot with a single unmodified pot in 2 sites) in 2013. Pots were baited and submerged for 48 hours at a time.

(1) Roosenburg W.M. & Green J.P. (2000) Impact of a bycatch reduction device on diamondback terrapin and blue crab capture in crab pots. *Ecological Applications*, 10, 882–889.

(2) Lowry M.B., Pease B.C., Graham K. & Walford T.R. (2005) Reducing the mortality of freshwater turtles in commercial fish traps. *Aquatic Conservation: Marine and Freshwater Ecosystems,* 15, 7–21.

(3) Fratto Z.W., Barko V.A., Pitts P.R., Sheriff S.L., Briggler J.T., Sullivan K.P., McKeage B.L. & Johnson T.R. (2008) Evaluation of turtle exclusion and escapement devices for hoop-nets. *Journal of Wildlife Management*, 72, 1628–1633.

(4) Fratto Z.W., Barko V.A. & Scheibe J.S. (2008) Development and efficacy of a bycatch reduction device for Wisconsin-type fyke nets deployed in freshwater systems. *Chelonian Conservation and Biology*, 7, 205–212.

(5) Rook M.A., Lipcius R.N., Bronner B.M. & Chambers R.M. (2010) Bycatch reduction device conserves diamondback terrapin without affecting catch of blue crab. *Marine Ecology Progress Series*, 409, 171–179.

(6) Hart K.M. & Crowder L.B. (2011) Mitigating by-catch of diamondback terrapins in crab pots. *The Journal of Wildlife Management*, 75, 264–272

(7) Larocque S.M., Cooke S.J. & Blouin-Demers G. (2012) Mitigating bycatch of freshwater turtles in passively fished fyke nets through the use of exclusion and escape modifications. *Fisheries Research*, 125, 149–155.

(8) Cairns N.A., Stoot L.J., Blouin-Demers G. & Cooke S.J. (2013) Refinement of bycatch reduction devices to exclude freshwater turtles from commercial fishing nets. *Endangered Species Research*, 22, 251–261.

(9) McKee R.K., Cecala K.K. & Dorcas M.E. (2016) Behavioural interactions of diamondback terrapins with crab pots demonstrate that bycatch reduction devices reduce entrapment. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 26, 1081–1089.

(10) Corso A.D., Huettenmoser J.C., Trani O.R., Angstadt K., Bilkovic D.M., Havens K.J., Russell T.M., Stanhope D. & Chambers R.M. (2017) Experiments with by-catch reduction devices to exclude diamondback terrapins and retain blue crabs. *Estuaries and Coasts*, 40, 1516–1522.

(11) Chavez S. & Williard A.S. (2017) The effects of bycatch reduction devices on diamondback terrapin and blue crab catch in the North Carolina commercial crab fishery. *Fisheries Research,* 186, 94–101.

Snakes & lizards

- **One study** evaluated the effects of installing exclusion devices on fishing gear on snake and lizard populations. This study was in Australia[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (0 STUDIES)

OTHER (1 STUDY)

- **Unwanted catch (1 study):** One replicated study off the coast of Western Australia[1] found that exclusion grids did not prevent sea snakes from entering trawl nets.

A replicated study in 2012 in off the coast of Western Australia (1) found that exclusion grids with escape hatches ('bycatch reduction device') did not prevent sea snakes from entering the codend of trawl nets in a tropical teleost fishery. In total 331 of 351 sea snakes passed through the exclusion grid, however only 16 sea snakes were recorded as trawl catch. The authors note that sea snakes were observed escaping through the trawl net and may have done so after passing through the grids. Loss of commercially-targeted teleost species from all trawls was 1% of catch. In June–December 2012, catch (target and unwanted) from three commercial trawl vessels was monitored using in-net and onboard cameras during daylight. Vessels were fitted with either: upward-facing grid and escape hatch with diamond-mesh net (372 trawl hours on 2 vessels), downward-facing grid and escape hatch with diamond-mesh net (559 trawl hours on 2 vessels), or downward-facing grid and escape hatch with square mesh net (389 trawl hours on 1 vessel; see original paper for all specifications). Use of bycatch reduction grids with escape hatches was mandatory in this fishery from 2006.

(1) Wakefield C.B., Santana-Garcon J., Dorman S.R., Blight S., Denham A., Wakeford J., Molony B.W. & Newman S.J. (2017) Performance of bycatch reduction devices varies for chondrichthyan, reptile, and cetacean mitigation in demersal fish trawls: assimilating subsurface interactions and unaccounted mortality. *ICES Journal of Marine Science*, 74, 343–358.

Crocodilians

- We found no studies that evaluated the effects of installing exclusion devices on fishing gear on crocodilian populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

6.29. Install escape devices on fishing gear

Background

Aquatic reptiles may become trapped or entangled in fishing nets (e.g. trawl nets, hoop nets, fyke nets) and traps. Escape devices such as escape holes, sections of larger mesh, or chimneys may be fitted to fishing gear to allow non-target animals to escape after they have entered fishing gear.

Due to the number of studies found, this action has been split by species group.

For studies that discuss the effect of using devices that exclude reptiles from fishing gear, see *Install exclusion devices on fishing gear*, and for studies on the effect of using a combination of exclusion and escape devices see *Install exclusion and escape devices on fishing gear.*

Sea turtles

- **One study** evaluated the effects of installing escape devices on fishing gear on sea turtle populations. This study was in the Gulf of Carpentaria[1] (Australia).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (0 STUDIES)

OTHER (1 STUDY)

- **Unwanted catch (1 Study):** One randomized, paired, controlled study in the Gulf of Carpentaria[1] found that trawl nets with escape devices caught a similar number of sea turtles compared to unmodified nets.

A randomized, paired, controlled study in 1995–1995 in seabed areas in the Gulf of Carpentaria, northern Australia (1) found that trawl nets fitted with one of seven escape zone designs ("bycatch reduction devices") caught similar numbers of sea turtles compared to unmodified nets. No statistical tests were carried out. Nets fitted with escape zones caught turtles at a similar rate (0.14 turtles/tow, 17 individuals) as unmodified nets (0.13 turtles/tow, 9 individuals). The unwanted catch included three species of turtles and three of snakes. The effect of escape zones on the commercially targeted prawn catch varied by design (see original paper for details). Escape zone designs tested included 'fisheye', 'radial escape section', 'square mesh window' and square mesh windows fitted with a number of modifications (see original paper for details). Vessels towed twin Florida Flyer prawn trawl nets from each side of the vessel in trials of one-month duration (sea turtles: February and October 1995). Nets fitted with one of the designs of escape zone and an unmodified net were randomly assigned to either side of the vessel.

(1) Brewer D., Rawlinson N., Eayrs S. & Burridge C. (1998) An assessment of bycatch reduction devices in a tropical Australian prawn trawl fishery. *Fisheries Research*, 36, 195–215.

Tortoises, terrapins, side-necked & softshell turtles

- **Three studies** evaluated the effects of installing escape devices on fishing gear on tortoise, terrapin, side-necked & softshell turtle populations. One study was in each of Australia[1], the USA[2] and Canada[3].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Survival (1 study):** One replicated, randomized, paired, controlled study in the USA[2] found that a lower percentage of turtles died in hoop nets with escape devices than in unmodified nets.

BEHAVIOUR (0 STUDIES)

OTHER (3 STUDIES)

- **Unwanted catch (3 Studies):** One replicated, controlled study in Australia[1] found that most short-necked turtles escaped from a carp trap with an escape ring. One replicated, randomized, controlled, paired study in the USA[2] found that hoop nets with escape devices caught fewer turtles than unmodified nets. One replicated, randomized, paired, controlled study in Canada[3] found that more painted turtles escaped from fyke nets with an escape device than from unmodified nets after being placed in the net.

A replicated, controlled study (years not provided) in a pool, lake and creek in New South Wales, Australia (1) found that adding an escape ring to a carp *Cyprinus carpio* trap allowed most short-necked turtles *Emydura macquarii* to escape. In escape trials, 85 of 120 turtles (71%) escaped within 90 minutes and 92 of 120 turtles (77%) escaped within four hours. Smaller turtles (average straight carapace width 17 cm) were more likely to escape than larger turtles (average width 19 cm). The average time for escapes was 63 minutes for centre-placed exits and 92 minutes for end placed exits. Very few carp escaped through the turtle exit during escape trials (14 of 120, 12% of fish escaped) and the authors reported that numbers of carp caught/day indicated that few carp were escaping through the turtle exit in fishing trials (see original paper). Cylindrical (90 cm diameter x 170 cm long) mesh carp traps were used modified with a 23 cm escape ring on the upper trap surface, either in the centre or at the opposite end to the entrance (which was closed for the experiment). A mesh platform was placed under the escape ring to aide turtles exiting. Ten individually-marked turtles were randomly selected to take part in six trials of each trap type. Turtles were placed as a group in the trap, submerged for 4 h and escapes recorded. The traps were also tested for carp escapes in a lake (escape trials) and creek (fishing trials).

A replicated, randomized, paired, controlled study in 2006 along a river in Missouri, USA (2) found that hoop nets modified with either a chimney or loose-weave mesh escape device caught fewer turtles than unmodified hoop nets in a catfish fishery. Modified hoop nets caught fewer turtles (chimney: 27 turtles caught in 13 of 49 nets, loose-weave mesh: 27 in 17 of 50 nets) than unmodified nets (166 in 33 of 50 nets). Thirteen of 27 turtles (48%) died in chimney-modified nets and 11 of 26 turtles (42%) died in loose-weave mesh modified nets compared to 101 of 166 turtles (61%) in unmodified nets (results were not statistically tested). Fewer target channel catfish *Ictalurus punctatus* were caught in modified nets (chimney: 8 individuals, loose-weave: 4) compared to unmodified nets (44 individuals). Numbers of flathead catfish *Pylodictis olivaris* caught in chimney-modified nets were similar (11 individuals) but numbers caught in loose-weave mesh modified nets were lower (1) than numbers caught in unmodified nets (26). Unmodified hoop nets (six hoops, 90 cm maximum hoop with 38 mm mesh, 3.7 m long) and hoop nets modified to allow turtles to escape with the addition of a section of larger loose-weave mesh or an escape chimney were deployed in pairs along four river stretches in May–July 2006 (50 unmodified nets, 49 chimney nets, 50 loose-weave nets) using a randomized block design. Nets were set for 48 hours at a time over nine weeks. The catch of turtle and commercially targeted catfish species was recorded.

A replicated, randomized, paired, controlled study in 2010–2011 in a freshwater lake in eastern Ontario, Canada (3) found that adding an escape chimney to fyke nets increased the escape rate of painted turtles *Chrysemys picta* and reduced the escape rate of fish compared to modifying nets with a large hole. More painted turtles escaped from fyke nets modified with escape chimneys (10 of 10 turtles escaped) compared to fyke nets modified with a large hole (12 of 20 turtles escaped). The proportion of fish escaping was reduced in escape-chimney nets (0.13 fish/24 hr) compared to large-hole nets (0.77 fish/24 hr). Escape rates of turtles and fishes were tested in modified commercial seven-hooped fyke nets set in a shallow warmwater lake (788 ha, nets set to 1.5 m depths) in April–June 2010–2011. Two nets were modified with either an open-topped chimney (a mesh tube 15 cm wide, 28 cm long and 28 cm tall) attached to the net between the 6th and 7th hoop (see original paper for details), or a large hole in the top (15 cm x 28 cm, typical of damage that occurs through normal fishing). Individual male painted turtles or fish *Lepomis* spp. were placed in the cod-end of a closed net for four hours (turtles) or 24 hours (fish) and escapes counted (turtles: 10 chimney trials and 20 large hole trials; fish: 10 chimney trials and 10 large hole trials).

(1) Lowry M.B., Pease B.C., Graham K. & Walford T.R. (2005) Reducing the mortality of freshwater turtles in commercial fish traps. *Aquatic Conservation: Marine and Freshwater Ecosystems,* 15, 7–21.

(2) Fratto Z.W., Barko V.A., Pitts P.R., Sheriff S.L., Briggler J.T., Sullivan K.P., McKeage B.L. & Johnson T.R. (2008) Evaluation of turtle exclusion and escapement devices for hoop-nets. *Journal of Wildlife Management*, 72, 1628–1633.

(3) Larocque S.M., Cooke S.J. & Blouin-Demers G. (2012) Mitigating bycatch of freshwater turtles in passively fished fyke nets through the use of exclusion and escape modifications. *Fisheries Research*, 125, 149–155.

Snakes & lizards

- **Three studies** evaluated the effects of installing escape devices on fishing gear on snake and lizard populations. All three studies were in the Gulf of Carpentaria[1,2,3] (Australia).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (0 STUDIES)

OTHER (3 STUDIES)

- **Unwanted catch (3 Studies):** One of two paired, controlled studies (including one randomized and one replicated study) in the Gulf of Carpentaria[1,2] found that trawl nets with escape devices caught a similar number of sea snakes compared to unmodified nets[1]. The other study[2] found that trawl nets with an escape device caught fewer sea snakes compared to unmodified nets. One replicated, paired, controlled study in the Gulf of Carpentaria[3] found that the placement of escape devices trawl nets affected the number of sea snakes caught compared to unmodified nets.

A randomized, paired, controlled study in 1995 in seabed areas in the Gulf of Carpentaria, northern Australia (1) found that trawl nets fitted with one of seven escape zone designs ("bycatch reduction devices") caught similar numbers of sea snakes compared to unmodified nets. No statistical tests were carried out. Nets fitted with escape zones caught sea snakes at a similar rate as unmodified nets (escape zones: 0.5 snakes/tow, 7 individuals; unmodified: 0.4 snakes/tow, 15 individuals). The unwanted catch included three species of snakes. The effect of escape zones on the commercially targeted prawn catch varied by design (see original paper for details). Escape zone designs tested included 'fisheye', 'radial escape section', 'square mesh window' and square mesh windows fitted with a number of modifications (see original paper for details). Vessels towed twin Florida Flyer prawn trawl nets from each side of the vessel in scientific trials of one-month duration (sea snakes: October 1995). Nets fitted with one of the designs of escape zone and an unmodified net were randomly assigned to either side of the vessel.

A replicated, paired, controlled study in 2002–2005 on the sea bottom in the Gulf of Carpentaria, Australia (2) found that adding a metal-barred escape hatch ('Yarrow Fisheye') to a prawn trawl net reduced unwanted catch of sea snakes. Overall, trawl nets modified with a metal-barred escape hatch caught 44% fewer sea snakes (76 snakes caught in 113 trawls) than unmodified trawl nets (134 snakes in 113 trawls). In separate trawls, catch rates of commercially-targeted tiger prawns *Penaeus* spp. were similar when modified (13–18 kg/net) and unmodified (13–19 kg/net) nets were used. Unwanted catch of sea snakes was assessed on a single commercial prawn trawler in September–November 2004 (41 trawls) and August–November 2005 (72 trawls). On each trawl, the vessel was fitted with a pair of nets (one starboard, one portside) both fitted with a metal-barred escape hatch in the codend (see original paper for design details) behind a downward-facing grid with escape zone ('Super Shooter' turtle excluder device). On each trawl, the escape hatch on one net was sewn shut (classed as unmodified) and the other was left open (classed as modified). The modified net was swapped between the starboard and port-side every two weeks by opening and sewing shut the escape holes on the nets in rotation. Prawn catch rates were assessed during 42 trawls over 13 nights in November 2002.

A replicated, paired, controlled study in 2004–2006 in benthic waters in the Gulf of Carpentaria, Australia (3) found that adding escape hole devices ('bycatch reduction device') ≤70 meshes from the codend of trawl nets in reduced unwanted catch of sea snakes compared to unmodified nets in a prawn fishery. Nets modified with escape hole devices located 30–70 meshes from the codend caught fewer sea snakes (82–168 snakes/trawl) compared to unmodified nets (99–350). Nets modified with devices located 120 meshes from the codend caught similar numbers of sea snakes (148–418 snakes/trawl) to unmodified nets (155–430). Unwanted catch of sea snakes was similar between three different escape hole devices (see paper for individual device details). Catch of commercially targeted prawns *Penaeus* spp. was similar between modified and unmodified nets, regardless of the location of the escape hole (see paper for details). In August–November 2004–2006, nets on commercial trawlers (10–15 trawlers/year) were modified with an oval framed 'fish eye' (930 trawls), a square-mesh panel (435 trawls), or a square opening with metal funnel below a rigid frame ('popeye' Fishbox design, 54 trawls) located 30–120 meshes from the codend. Trawlers fished pairs of modified and unmodified nets, one on either side of the boat (the modified net was switched sides approximately fortnightly). Turtle excluder devices (frames in front of the codend with escape holes) were mandatory and used on all nets. Crew and independent scientific observers identified sea snakes landed with catch.

(1) Brewer D., Rawlinson N., Eayrs S. & Burridge C. (1998) An assessment of bycatch reduction devices in a tropical Australian prawn trawl fishery. *Fisheries Research*, 36, 195–215.
(2) Heales D.S., Gregor R., Wakeford J., Wang Y.G., Yarrow J. & Milton D.A. (2008) Tropical prawn trawl bycatch of fish and seasnakes reduced by Yarrow Fisheye Bycatch Reduction Device. *Fisheries Research*, 89, 76–83.
(3) Milton D.A., Fry G.C. & Dell Q. (2009) Reducing impacts of trawling on protected sea snakes: By-catch reduction devices improve escapement and survival. *Marine and Freshwater Research*, 60, 824–832.

Crocodilians

- We found no studies that evaluated the effects of installing escape devices on fishing gear on crocodilian populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

6.30. Install exclusion and escape devices on fishing gear

- **Six studies** evaluated the effects of installing exclusion and escape devices on fishing gear on reptile populations. Two studies each were off the coast of Australia[1,2], in the Gulf of Carpentaria[3,4] (Australia) and in the Adriatic Sea[5,6].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (2 STUDIES)

- **Use (2 studies):** Two replicated studies (including one controlled study) in the Adriatic Sea[5,6] found that one[5] or two[6] loggerhead turtles were able to escape from a trawl net with an exclusion and escape device.

OTHER (5 STUDIES)

- **Unwanted catch (5 studies):** Four studies (including two replicated, paired, controlled studies) off the coast of Australia[1,2] and in the Gulf of Carpentaria[3,4] (Australia) found that that trawl nets with an exclusion and escape device caught fewer loggerhead turtles[1,2] or sea turtles and sea snakes[3,4] compared to unmodified nets. One replicated study in the Adriatic Sea[5] found that no loggerhead turtles were caught by a trawl net with an exclusion and escape device.

Background

Aquatic reptiles may become trapped or entangled in fishing nets (e.g. trawl nets, hoop nets, fyke nets) and traps. Exclusion devices, such as grids, mesh, funnels, rings, or rectangular inserts, can be installed in an attempt to reduce the number of non-target animals that are caught. Escape devices such as escape holes, sections of larger mesh, or chimneys may be fitted to fishing gear to allow non-target animals to escape after they have entered fishing gear. Here we include studies that test the effect of a combination of exclusion and escape devices.

For studies that look at these actions separately, see *Install exclusion devices on fishing gear* and *Install escape devices on fishing gear*.

A replicated, controlled study in 1991–1992 in oceanic and estuarine waters off the coast of Queensland, Australia (1) found that when a soft mesh panel with escape hole ('Morrison soft TED') was added to a trawl net no loggerhead turtles *Caretta caretta* were caught accidentally. No loggerhead turtles were caught in the modified net but were occasionally caught in the unmodified net (no data on turtles are provided). Catch rates of target prawns *Penaeus* spp. and *Metapenaeus bennettae* varied between no significant reduction and a 29% reduction in modified compared to unmodified nets, depending on location and season (data reported as a cross-site analysis of fishing power, see original paper for details). Trawl nets modified by adding a polypropylene mesh panel with an escape hole in front of the codend were tested in two trials (May 1991 and January 1992) on a 15 m research trawler in an oceanic site with sandy substrate and an estuarine site with a muddy bottom. Unmodified nets were towed first (number of trawls not specified) and then the modified net was used. Between 17 and 23 tows were completed in each trial, each lasting 45–100 minutes.

A replicated, paired, controlled study (years not provided) in five oceanic-sand and estuarine-silt-bottomed sites in the coast off of southeastern Queensland, Australia (2) found that adding a device that included a funnel to direct unwanted catch, an upward-facing flexible grid and covered escape panel ("ausTED" design turtle excluder device) onto a trawl net reduced the catch of loggerhead turtles *Caretta caretta* compared to a standard net. Using the device reduced the catch of loggerhead turtles (0 turtles) compared to nets without the device (7 turtles). In a separate experiment, six juvenile green turtles *Chelonia mydas* were placed in the path of nets with the device. Only three passed into the net entrance and all were successfully excluded from the main part of the net. Commercially-targeted prawn *Penaeus* spp. and *Metapenaeus* spp. catch rates, size and quality were similar in nets with and without the device (see original paper for details). Two commercial 6.8 m long trawl nets (40 mm mesh) were attached to a 15 m trawler. The device was fitted to one of the nets. Between 13 and 27 linear tows were conducted per site (each 60 minutes long, 85 tows used the device in total). Juvenile green turtles placed in the path of the net were videoed to assess how the device assisted their escape.

A randomized, paired, controlled study in 1995 in seabed areas in the Gulf of Carpentaria, northern Australia (3) found that trawl nets fitted with one of three different grids accompanied by an escape hole ("turtle excluder device") and a secondary escape zone ("bycatch reduction device") reduced the catch of sea turtles and sea snakes, compared to unmodified nets. No statistical tests were carried out. Nets fitted with both a turtle excluder device and a bycatch reduction device caught fewer turtles (0.005 turtles/tow, 1 individual) compared to unmodified nets (0.10 turtles/tow, 11 individuals). Overall, nets fitted with both a turtle excluder device and a bycatch reduction device caught fewer sea snakes (0.36 snakes/tow, 45 individuals), than unmodified nets (0.42 snakes/tow, 15 individuals). However, some combinations of a turtle excluder device and a bycatch reduction device caught fewer sea snakes (flexible upward grid with fisheye in front of grid: 0.2 snakes/tow, 3 individuals; square upward grid with square mesh: 0.2 snakes/tow, 8 individuals) compared to unmodified nets or other combinations tested (see original paper for details). The unwanted catch included three species of turtles and three of snakes. The effect of grids and escape zones on the commercially targeted prawn catch varied by design (see original paper for details). Devices tested included a flexible, circular, upward tilted grid with top escape hole ('AusTED') and a secondary escape zone in front of the grid; a square, upward tilted grid with a top escape hole ('Nordmøre') and secondary escape zone after the grid; and a circular downward tilted grid ('Super Shooter') with a secondary escape zone after the grid. Secondary escape zones included different configurations of a 'fish eye' or 'square mesh window' (see original paper for full details). Vessels towed twin Florida Flyer prawn trawl nets from each side of the vessel in scientific trials of one-month duration (sea turtles: February 1995, October 1995, October 1996; sea snakes: October 1995). Nets fitted with one of the designs of grid and an unmodified net were randomly assigned to either side of the vessel.

A replicated, paired, controlled study in 2001 in areas of seabed in the Gulf of Carpentaria, northern Australia (4) found that nets fitted with a mesh escape window ("bycatch reduction device") and a grid ("turtle excluder device") caught fewer sea turtles and sea snakes, compared to unmodified nets. Nets fitted with both an escape window and grid caught 100% fewer sea turtles (with devices: 0 turtles; unmodified nets: 66 turtles) and 5% fewer sea snakes (number not provided), compared to unmodified nets. The use of a "turtle excluder device" and a "bycatch reduction device" had been compulsory since 2000 in the Australian prawn fishery. Commercial vessels towed twin Florida Flyer prawn trawl nets from each side of the vessels in August–November 2001. Modified nets were fitted with one of two designs of escape window (a "Bigeye" design or a square-mesh escape window) and either an upward or downward facing exclusion grid (rigid or semi-rigid frame with ≤120 mm bar spacing and an opening of ≥700 mm). A modified and an unmodified net were randomly assigned to either side of the vessel and towed simultaneously (33 modified nets examined for sea turtles, 214 for sea snakes; 84 unmodified nets for sea turtles, 432 for sea snakes). The combinations of various device designs were not compared. Where possible, sea turtles (4 species) and sea snakes (12 species) caught were identified to species. The "Bigeye" design was later removed from the Australian list of approved escape zone designs.

A replicated study in 2008 on the sea bottom in the Adriatic Sea (5) found that adding a downward-facing grid and bottom escape hole ('Super Shooter' model of 'turtle excluder device') to standard trawl nets allowed a loggerhead turtle *Caretta caretta* to escape after being caught. No statistical tests were carried out for unwanted catch. During trials, one turtle entered a trawl net modified with a 'Super Shooter' and was successfully excluded. No turtles entered trawl nets modified with two other excluder devices that were tested. The 'Super Shooter' retained the most catch (20 kg/tow) and had the lowest discards (9 kg/tow) of commercially-targeted European hake *Merluccius merluccius*, compared to the two other excluder devices that were tested (retention rate: 13–18 kg/tow; discard rate: 12–21 kg/tow). All turtle excluder devices were downward-facing grids (set to an angle of 45–48 degrees) located immediately in front of the codend and accompanied by a bottom escape hole. Four different excluder devices were tested: a lightweight rigid aluminium grid (which broke down and was excluded from the study); a flexible mixed-cable grid; a semi-rigid grid of steel and rubber; a 'Super Shooter' aluminium grid with enlarged space between bars (see original paper for details). Data were collected during 42 tows with an average duration of 48 minutes (11–15 tows/excluder device). Excluder devices were tested on a four-sided net using standard commercial trawl fishing rigging and operation.

A replicated, controlled study in 2014 on muddy-sandy seabed in the northern Adriatic Sea (6) found that using an upward-facing flexible grid with escape hole ('turtle excluder device') in a bottom-trawl net allowed loggerhead turtles *Caretta caretta* to escape after entering the net. No statistical tests were carried out. Nets with a flexible grid and escape hole allowed two loggerhead turtles *Caretta caretta* to escape¸ while one turtle was caught in an unmodified net. Commercial fish catch was similar between modified and unmodified nets (modified net: 8–26 kg/hr; unmodified net: 10–34 kg/hr). Fifty-five bottom trawl trials were carried out (20–40 m depths) in spring, autumn and winter 2014. A total of 25 trawls used the experimental turtle excluder device and 30 used a traditional net as a control (nets were 58 m long). The turtle excluder device was a circular plastic grid set at an upward angle with a top escape hole (a net panel with three sides sewn onto the trawl net) installed front of the codend. Escapes from modified nets were monitored using an underwater camera.

(1) Robins-Troeger J.B. (1994) Evaluation of the Morrison Soft Turtle Excluder Device - Prawn and Bycatch Variation in Moreton Bay, Queensland. *Fisheries Research*, 19, 205–217.
(2) Robins-Troeger J.B., Buckworth R.C. & Dredge M.C.L. (1995) Development of a trawl efficiency device (TED) for Australian prawn fisheries. II. Field evaluations of the AusTED. *Fisheries Research*, 22, 107–117.
(3) Brewer D., Rawlinson N., Eayrs S. & Burridge C. (1998) An assessment of bycatch reduction devices in a tropical Australian prawn trawl fishery. *Fisheries Research*, 36, 195–215.
(4) Brewer D., Heales D., Milton D., Dell Q., Fry G., Venables B. & Jones P. (2006) The impact of turtle excluder devices and bycatch reduction devices on diverse tropical marine communities in Australia's northern prawn trawl fishery. *Fisheries Research*, 81, 176–188.
(5) Sala A., Lucchetti A. & Affronte M. (2011) Effects of Turtle Excluder Devices on bycatch and discard reduction in the demersal fisheries of Mediterranean Sea. *Aquatic Living Resources*, 24, 183–192.
(6) Lucchetti A., Punzo E. & Virgili M. (2016) Flexible Turtle Excluder Device (TED): an effective tool for Mediterranean coastal multispecies bottom trawl fisheries. *Aquatic Living Resources*, 29, 201.

6.31. Use sinking lines instead of floating lines

- We found no studies that evaluated the effects on reptile populations of using sinking lines instead of floating lines.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background
Using sinking or non-buoyant lines (e.g. between pots or traps) that lie closer to the sea floor or riverbed instead of floating in the water column may reduce the risk of marine and freshwater reptiles becoming entangled.

6.32. Use stiffened materials or increase tension of fishing gear

- We found no studies that evaluated the effects on reptile populations of using stiffened materials or increasing tension of fishing gear.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background
Using stiffened materials or increasing the tension of fishing nets, ropes or lines may reduce the risk of reptiles becoming entangled.

6.33. Modify mesh sizes used in fishing gear

- We found no studies that evaluated the effects on reptile populations of modifying mesh sizes used in fishing gear.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background
Numerous fishing methods involve the use of panels of meshed netting to harvest the target species, and reptiles may become entangled in these nets. Modifying mesh sizes may reduce the chance of reptiles becoming entangled.

6.34. Use lower profile gillnets with longer/no tie-downs

- We found no studies that evaluated the effects on reptile populations of using lower profile gillnets with longer/no tie-downs.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background
Gillnets hang in the water at various depths and may entangle reptiles. Tie-downs are lines that are shorter than the height of the net and connect the floatline (top) and leadline (bottom), causing the net to billow out, potentially increasing the chance of entanglement. Lower profile (narrower) nets without tie-downs may reduce the number of reptiles that become entangled.

6.35. Use bindings to keep trawl nets closed until they have sunk below the water surface

- We found no studies that evaluated the effects on reptile populations of using bindings to keep trawl nets closed until they have sunk below the water surface.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background
Bindings may be used to keep trawl nets closed until they have sunk below the water surface. This may reduce the risk of aquatic reptiles becoming entangled.

Bait

6.36. Use dyed bait

- **Two studies** evaluated the effects of using dyed bait on reptile populations. One study was in Costa Rica[1] and one was in the North Pacific[2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One randomized, paired, controlled study in Costa Rica[1] found that loggerhead and Kemp's ridley turtles showed mixed preferences for dyed compared to non-dyed bait in captive trials.

OTHER (2 STUDIES)

- **Unwanted catch (2 studies):** Two paired studies (including one randomized, controlled study) in Costa Rica[1] and the North Pacific[2] found that hooks with dyed bait caught a similar number of olive ridley and green turtles[1] and loggerhead turtles[2] compared to hooks with non-dyed bait.

Background
Changing the colour of bait may reduce the attractiveness of bait to reptiles. This may reduce reptile captures and entanglements in fishing gear, and in doing so reduce the loss of catch for fishers.

See also: *Use a different bait type* and *Change hook baiting technique*.

A randomized, paired, controlled study in 2001–2003 in pelagic waters in the Gulf of Papagayo, Costa Rica (1) found that using blue-dyed bait in a longline fishery did not reduce unwanted catch of olive ridley turtles *Lepidochelys olivacea* and green turtles *Chelonia mydas agassizi*, and in separate captive trials found that preference for dyed or non-dyed bait varied depending on the turtle species. Turtle catch rates were similar for blue bait (8 turtles/1,000 hooks, 13 individuals) and non-dyed bait (8 turtles/1,000 hooks, 9 individuals). In separate captive trials, loggerhead turtles *Caretta caretta* preferred non-dyed bait to blue or red bait and Kemp's ridley turtles *Lepidochelys kempii* preferred non-dyed bait to blue bait but red bait to non-dyed bait (see original paper for details). Field trials were carried out simultaneously on two commercial longline fishing vessels in December 2013. In total, 22 lines were deployed using circle hooks (size: 12/0, 560–606 average hooks/deployment, each deployment lasted 8 hours) and baited with blue-dyed (9 deployments) or non-dyed (13 deployments) bait, which was either squid *Loligo* spp. (12 deployments) or sailfish *Istiophorus platypterus* (10 deployments). Captive trials tested preferences of two-year-old, captive reared loggerhead (four trials in October 2001–March 2002, 49 individuals) and olive ridley turtles (one trial in July-August 2002, 42 individuals) for dyed (red or blue) compared to non-dyed squid pieces placed in pools (see original paper for details).

A paired study in 2002–2003 in pelagic waters in the North Pacific Ocean (2) found that using blue–dyed fishing bait did not reduce unwanted catch of loggerhead turtles *Caretta caretta* in a longline fishery, regardless of bait species used. There was no difference in the rates of unwanted catch of loggerhead turtles between blue-dyed and non-dyed bait (9 individuals caught in each case). All turtles were caught alive and subsequently released. Bait colour did not alter the catch rates of commercially-targeted swordfish *Xiphias gladius*. Longlines were deployed from a single vessel (54 m long) in May–June 2002 and 2003 (19 deployments/year). Whole mackerel *Scomber japonicus* and squid *Todarodes pacificus* were used as bait. Half of the bait of both species was dyed blue using non-toxic dye and the two bait species (dyed and non-dyed) were attached alternately to standard Japanese hooks (size 3.8-sun, 10° offset; 960 total hooks). Hooks were set to a depth of 40–90 m and lines were deployed overnight.

(1) Swimmer Y., Arauz R., Higgins B., McNaughton L., McCracken M., Ballestero J. & Brill R. (2005) Food color and marine turtle feeding behavior: Can blue bait reduce turtle bycatch in commercial fisheries? *Marine Ecology Progress Series*, 295, 273–278.

(2) Yokota K., Kiyota M. & Okamura H. (2009) Effect of bait species and color on sea turtle bycatch and fish catch in a pelagic longline fishery. *Fisheries Research*, 97, 53–58.

6.37. Use a different bait type

Background

Using alternative bait that is less attractive to marine and freshwater reptiles may reduce entanglement and capture of reptiles in fishing gear. Losses to fishers may also be reduced thereby reducing human-wildlife conflict.

Due to the number of studies found, this action has been split by species group.

See also: *Use dyed bait* and *Change hook baiting technique*.

Sea turtles

- **Nine studies** evaluated the effects of using a different bait type on sea turtle populations. Three studies were in each of the Atlantic[1,6,7] and Pacific[2,3,8], and one was in each of the Atlantic and north Pacific[9], the Gulf of Garbes[4] (Tunisia) and Italy[5].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Survival (2 studies):** Two studies (including one replicated, controlled study) off the coast of Hawaii[2] and in the Southern Atlantic[6] found that the percentage of loggerhead and leatherback turtles that survived being caught by fish-baited or squid-baited hooks[6] or fish-baited circle hooks and squid-baited J-hooks[2] was similar.
- **Condition (1 study):** One before-and-after study off the coast of Hawaii[2] found that fish-baited circle hooks deeply hooked fewer leatherback and hard-shell turtles compared to squid-baited J-hooks.

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One controlled study in Italy[5] found that loggerhead turtles in a captive setting were less likely to bite at fish bait than squid bait. The study[5] also found that smaller turtles were more likely to bite at mackerel bait and larger turtles at squid bait.

OTHER (8 STUDIES)

- **Unwanted catch (8 studies):** Four of five studies (including one replicated, paired, controlled study) in the North Pacific[3], Eastern Pacific[8], Atlantic[6,7] and Atlantic and North Pacific[9] found that fish-baited hooks caught fewer sea turtles[3,6,9] or were swallowed by fewer olive ridley turtles[8] than squid baited hooks. One study[8] also found that fish bait in combination with larger circle hooks lead to the highest percentage of external hookings. The other study[7] found mixed effects of using fish or squid-baited hooks on the unwanted catch of hard-shell and leatherback turtles. One replicated, controlled study in the north-western Atlantic Ocean[1] found that fish-baited J-hooks caught fewer sea turtles compared to squid-baited hooks. The study[1] also found that unwanted catch was more similar for fish-baited and squid-baited circle hooks. One before-and-after study off the coast of Hawaii[2] found that fish-baited circle hooks caught fewer loggerhead and leatherback turtles compared to compared to squid-baited J-hooks. One replicated study in the Gulf of Garbes[4] found that hooks baited with stingray caught fewer loggerhead turtles compared to fish-baited hooks.

A replicated, controlled study in 2002 in pelagic waters in the north-western Atlantic Ocean (1) found that unwanted catch of sea turtles was reduced when using mackerel-baited *Scomber scombrus* instead of squid-baited *Illex* spp. J-hooks, and tended to be lower when circle hooks were used in a tuna *Thunnus* spp. and swordfish *Xiphias gladius* longline fishery. Unwanted catch of sea turtles was reduced when mackerel-baited J-hooks were used (0.13–0.15 turtles/1,000 hooks) compared to squid-baited J-hooks (0.5 turtles/1,000 hooks). When mackerel-baited circle hooks were used, unwanted catch was 0.04–0.15 turtles/1,000 hooks compared to 0.05–0.21 turtles/1,000 hooks when squid-baited circle hooks were used (results were not statistically tested, see original paper for details including individual species responses). Commercially-targeted swordfish catch increased when mackerel was used (see original paper). Five hook/bait combinations were trialled: (1) 0° offset 18/0 circle hooks with 150–300 g squid bait, (2) 10° offset 18/0 circle hooks with squid bait, (3) 20°–25° offset 9/0 J-hooks with 200–500 g mackerel bait, (4) 10° offset 18/0 circle hooks with mackerel bait and (5) 20°–25° offset 9/0 J-hooks with squid bait (standard in the fishery). Thirteen vessels made 489 deployments, fishing a total of 427,382 hooks

(71,000 hooks for combinations 1–4 and 142,000 hooks for combination 5). Onboard observers collected catch data.

A before-and-after study in 1994–2006 in pelagic waters off the coast of Hawaii, USA (2) found fish-baited circle hooks reduced unwanted catch of sea turtles compared to squid-baited J-hooks in a swordfish *Xiphias gladius* longline fishery. Capture rates of leatherback turtles *Dermochelys coriacea* reduced by 83% (0.006 turtles/1,000 hooks) and loggerhead turtles *Caretta caretta* by 90% (0.012 turtles/1,000 hooks) when fish-baited circle hooks were used compared to squid-baited J-hooks (leatherback: 0.03 turtles/1,000 hooks, loggerhead: 0.13 turtles/1,000 hooks). Mortality rates were similar whether fish-baited circle or squid-baited J hooks were used (circle: 35 of 35 turtles survived, J: 180 of 182). Fewer turtles were deeply hooked when fish-baited circle hooks were used (leatherback: 0%, hard-shell: 22%) compared to squid-baited J hooks (10%, 60%). Swordfish catch increased by 16% after fish-baited circle hooks were introduced, but tuna (Scombridae), mahi mahi *Coryphaena* spp, opah *Lampris* spp. and wahoo *Acanthocybium solandri* catch reduced by 34–50% (see paper for details). Catch data from the US National Marine Fisheries Service observer programme were compared from before and after regulations were introduced requiring the use of 10° offset 18/0 circle hooks with fish bait in a pelagic swordfish longline fishery. Prior to the regulations, 9/0 J hooks with squid bait were used. 'Before' data used was from 1994–2002 (120 observed trips of 1,631 sets with 1,282,748 J hooks deployed) and 'after' data was from 2004–2006 (164 observed trips of 2,631 sets with 2,150,674 hooks deployed).

A paired study in 2002–2003 in pelagic waters in the North Pacific Ocean (3) found that using fish instead of squid as bait reduced unwanted catch of loggerhead turtles *Caretta caretta* in a shallow-set longline fishery. Catch of loggerhead turtles was reduced using mackerel *Scomber japonicus* bait (4 individuals) compared to using Japanese common squid *Todarodes pacificus* (18 individuals). All turtles were caught alive and subsequently released. Bait type did not alter the catch rates of commercially-targeted swordfish *Xiphias gladius*, bigeye tuna *Thunnus obesus*, blue shark *Prionace glauca*, and shortfin mako shark *Isurus oxrinchus* (see original paper for details), but increased catch of striped marlin *Tetrapturus audax* (mackerel: 14 individuals; squid: 5 individuals). Longlines were deployed from a single vessel (54 m long) in May-June 2002 and 2003 (19 deployments/year). Whole mackerel and squid were used as bait. The two bait species were attached alternately to standard Japanese hooks (size: 3.8-sun, 10° offset, 960 total hooks). Hooks were set to a depth of 40–90 m and lines were deployed overnight.

A replicated study in 2007–2008 in pelagic waters in the south of the Gulf of Garbes, Tunisia (4) found that using stingray *Dasyatis pastinaca* as bait in a longline fishery reduced unwanted catch of loggerhead turtles *Caretta caretta* compared to using mackerel *Scomber scombrus*, but also increased catch of commercially-targeted sharks. Fewer turtles were caught with stingray bait (0.2 turtles/1000 hooks) compared to mackerel (1.2). Catch of commercially targeted sandbar sharks *Carcharinus plumbeus* was higher with stingray bait (19 sharks/1,000 hooks) compared to mackerel (13 sharks/1,000 hooks). J-hooks (111 mm long, 57 mm wide) were baited with pieces of stingray or whole mackerel. Data were collected by onboard observers over 21 trips on longline vessels during July–September in 2007 and 2008. In total, 48 sets of fishing gear were deployed overnight (stingray: 19, mackerel 29 deployments) using 35,950 hooks (stingray: 13,800, mackerel: 22,150 hooks) during 21 fishing trips. Fishing gear comprised a mainline (20–35 km long) with branchlines (8 m long) suspended horizontally by floats. Baited J-hooks were located at the end of branchlines approximately 40 m apart.

A controlled study in 2001–2010 in sea water test tanks in Italy (5) found that loggerhead turtles *Caretta caretta* were less likely to bite at mackerel *Scomber scomber* bait than squid *Ilex argentinus* bait, and that this varied with the size of the turtle. Overall, turtles were less likely to bite at mackerel bait (biting at mackerel: 23% frequency) than squid (biting at squid: 60% frequency), but more likely to bite at mackerel than no bait (5–8% frequency). Smaller turtles were more likely to bite at mackerel bait and larger turtles were more likely to bite at squid bait (data reported as statistical model outputs). Whole mackerel and squid were selected as bait as these are commonly used in longline fisheries. Individual turtles (30 in total) were presented with bait of the same species (13 mackerel tests; 20 squid tests) and no bait in three different coloured sacks and their response was recorded on a portable video camera. Three turtles were tested using both bait types. Attempts to bite a sack were considered proof of biting behaviour. Turtles were wild caught individuals who had been in the rescue centre for <4 months and were considered fit to be released.

A replicated, controlled study in 2008–2012 in pelagic waters in the Southern Atlantic (6) found that using mackerel *Scomber* spp. bait reduced unwanted catch of loggerhead turtles *Caretta caretta* and leatherback turtles *Dermochelys coriacea* compared to using squid *Illex* spp., but when squid bait was used, unwanted catch rates depended on the hook type used. Unwanted turtle catch was lower when mackerel bait was used regardless of hook type (non-offset circle: 0.2 turtles/1,000 hooks; offset circle: 0.2; J-hook: 0.3), compared to squid bait. The number of turtles caught by squid bait varied with hook type (non-offset circle hooks: 0.7 turtles/1000 hooks; offset circle hooks: 0.6 turtles; J-hooks 1.7). This pattern was observed for both leatherback and loggerhead turtle species (see paper for details). Overall turtle mortality rates were similar regardless of whether squid (146/228 individuals alive) or mackerel (40/58 individuals alive) was used as bait. Three hook types baited with either squid or mackerel were used alternately on a commercial longline fishing vessel: traditional J-hook (size: 9/0) and two circle hooks (a non-offset and a 10^{o} offset, both sized: 17/0; 148,800 total hooks/type). In total 310 longline deployments (1,440 hooks/deployment; 446,400 total hooks, lines set to 20–50 m depths) were carried out overnight in October 2008–February 2012. One bait type was used in each deployment. Turtle catch was monitored by onboard observers.

A replicated, paired, controlled study in 2008–2011 in pelagic waters in the north-east Atlantic Ocean (7) found that changing from squid *Illex* spp. bait to mackerel *Scomber* spp. bait reduced unwanted catch of hard-shell sea turtles (Cheloniidae spp.), but not leatherback turtles *Dermochelys coriacea* in a pelagic longline swordfish *Xiphias gladius* fishery. Unwanted catch of hard-shell sea turtles (loggerhead *Caretta* caretta, olive ridley *Lepidochelys olivacea* and Kemp's ridley *Lepidochelys kempii* turtles) was reduced when mackerel bait was used (0.07–0.16 turtles/1,000 hooks) compared to squid (0.14–0.35 turtles/1,000 hooks). Unwanted catch of leatherback turtles was similar when mackerel (0.39–0.95 turtles/1,000 hooks) or squid (0.50–0.10 turtles/1,000 hooks) bait was used. In August 2008–December 2011, a commercial vessel carried out 202 overnight longline fishing deployments (lines: 55 nm long with 5 branchlines, deployed 20–50 m deep, lit by green lights). Whole squid or mackerel were used as bait (one type of bait/line deployment). Hook styles (10° offset J-hooks traditionally used in the fishery; non-offset G-style circle hooks; and 10° offset Gt-style circle hooks) were alternated every 70–80 hooks along the line in a randomized start order (254,520 total hooks deployed with 42,420 of each hook/bait combination). Unwanted catch was counted and released.

A replicated study in 2004–2011 in pelagic waters in the Eastern Pacific Ocean (8) found that using fish instead of squid bait reduced the likelihood of olive ridley turtles *Lepidochelys olivacea* swallowing hooks in an artisanal surface longline fishery. When fish bait was used, hooks were less likely to be swallowed by olive ridley turtles than when squid bait was used (results reported as odds ratios, see original paper for details). Using fish bait in combination with larger circle hooks lead to the largest proportion of external hookings. In 2004–2011, incidental catch rates of sea turtles on circle hooks (sizes 12/0–18/0), tuna hooks and traditional J-hooks (see original paper for hook details) were compared by placing hooks in alternative sequence along longlines (3.5 million total hooks used in 8,996 line deployments) in an artisanal longline fishery. Bait used was classed as squid (*Dosidicus gigas, Illex* sp. and *Loligo* sp.) or fish (*Opisthonema* spp., *Scomber japonicus, Auxis* spp. and *Sardinops sagax*) and only deployments using one type of bait were included in the analysis (4,838 of 8,996 line deployments). Information on hooking location and entanglement of sea turtles was recorded (1,823 total olive ridley turtles).

A replicated, before-and-after study in pelagic longline fisheries in the Atlantic and North Pacific (9) found that using fish bait resulted in less leatherback *Dermochelys coriacea* and loggerhead *Caretta caretta* turtle bycatch compared to when squid bait was used. The number of turtles caught on longlines was lower in the Atlantic when fish bait was used (leatherback: 0–6% chance with circle hooks, 13% with J hooks; Loggerhead: 0–5% with circle hooks, 9% with J hooks) compared to squid bait (leatherback: 9% with circle hooks, 20% with J hooks; Loggerhead: 11% with circle hooks, 18% with J hooks). The same was true in the Pacific (loggerhead: circle hook: 1% with fish, 2% with squid; j hook: 5% with fish, 13% with squid). Following the introduction of regulations on bait and hooks, overall turtle bycatch was reduced in both the Atlantic (leatherback: 40% reduction; loggerhead: 61% reduction) and Pacific (leatherback: 84% reduction; loggerhead 95% reduction). Fisheries were closed in 2001 and re-opened with regulations regarding bait (fish or squid) and hook type (circle or J hooks) (see paper for details). Pelagic Observer Program data from before (1992–2001) and after (2004–2015) regulations was used to determine the number of turtles caught/1,000 hooks.

(1) Watson J.W., Epperly S.P., Shah A.K. & Foster D.G. (2005) Fishing methods to reduce sea turtle mortality associated with pelagic longlines. *Canadian Journal of Fisheries and Aquatic Sciences* 62, 965–981.

(2) Gilman E., Kobayashi D., Swenarton T., Brothers N., Dalzell P. & Kinan-Kelly I. (2007) Reducing sea turtle interactions in the Hawaii-based longline swordfish fishery. *Biological Conservation*, 139, 19–28.

(3) Yokota K., Kiyota M. & Okamura H. (2009) Effect of bait species and color on sea turtle bycatch and fish catch in a pelagic longline fishery. *Fisheries Research*, 97, 53–58.
(4) Echwikhi K., Jribi I., Bradai M.N. & Bouain A. (2010) Effect of type of bait on pelagic longline fishery-loggerhead turtle interactions in the Gulf of Gabes (Tunisia). *Aquatic Conservation Marine and Freshwater Ecosystems*, 20, 525–530.
(5) Piovano S., Farcomeni A. & Giacoma C. (2012) Effects of chemicals from longline baits on the biting behaviour of loggerhead sea turtles. *African Journal of Marine Science*, 34, 283–287.
(6) Santos M.N., Coelho R., Fernandez - Carvalho J. & Amorim S. (2013) Effects of 17/0 circle hooks and bait on sea turtles bycatch in a Southern Atlantic swordfish longline fishery. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 23, 732 - 744.
(7) Coelho R., Santos M.N., Fernandez-Carvalho J. & Amorim S. (2015) Effects of hook and bait in a tropical northeast Atlantic pelagic longline fishery: Part I-Incidental sea turtle bycatch. *Fisheries Research*, 164, 302–311.
(8) Parga M.L., Pons M., Andraka S., Rendon L., Mituhasi T., Hall M., Pacheco L., Segura A., Osmond M. & Vogel N. (2015) Hooking locations in sea turtles incidentally captured by artisanal longline fisheries in the Eastern Pacific Ocean. *Fisheries Research*, 164, 231–237.
(9) Swimmer Y., Gutierrez A., Bigelow K., Barceló C., Schroeder B., Keene K., Shattenkirk K. & Foster D.G. (2017) Sea turtle bycatch mitigation in U.S. longline fisheries. *Frontiers in Marine Science*, 4, 260.

Tortoises, terrapins, side-necked & softshell turtles

- **Two studies** evaluated the effects of using a different bait type on tortoise, terrapin, side-necked and softshell turtles. Both studies were in the USA[1,2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (0 STUDIES)

OTHER (2 STUDIES)

- **Unwanted catch (2 studies):** One randomized, controlled study in the USA[1] found that a crab pot with mackerel bait caught more diamondback terrapins than when chicken bait or no bait was used. One replicated, paired study in the USA[2] found that hoop nets with soap bait caught fewer turtles than nets with cheese bait.

A randomized, controlled study (years not provided) in a brackish water experimental enclosure in South Carolina, USA (1) found that using mackerel bait in a crab pot increased catch rates of diamondback terrapin *Malaclemys terrapin* compared to chicken or no bait. Mackerel bait increased the number of terrapins caught (1.2 entries/terrapin/h) compared to chicken or no bait, which produced similar results (chicken: 0.6; no bait: 0.2 entries/terrapin/h). In total, 25 wild terrapins were caught to participate in three randomly ordered trials: mackerel bait, chicken bait and no bait. A single crab pot with chimney was used to test each bait type. Terrapins were monitored by webcam in 90-minute videos/treatment.

A replicated, paired study in 2014 of 13 reservoirs in Kentucky, USA (2) found that using soap rather than cheese as fishing bait in hoop nets reduced unwanted catch of turtles in a catfish *Ictalurus punctatus* fishery. Unwanted catch of all turtles in hoop nets was reduced with soap bait (7 turtles/net deployment) compared to cheese bait (11 turtles/net deployment). Turtle mortality was reduced with soap bait compared to cheese (data reported as statistical model outputs). Catch rates of commercially targeted catfish were similar between soap-baited (1,613 individuals) and cheese-baited hoop nets (1,429 individuals) although soap-baited nets caught larger catfish (344 mm average length) compared to cheese-baited (321 mm). In June 2014, four to six tandem hoop net combinations (three nets/combination, each 3.4 m long with 25 mm bar mesh and seven 0.8 m hoops) were deployed at <4 m depths in 13 reservoirs (70 total net deployments, two sampling periods). Nets were either baited with 800g cheese logs or 800g Zote © soap. Nets were fished for two days; all animals were removed and nets were then reset with the opposite bait and fished for a further two days. In total six turtle species were caught, of which three species (red-eared slider *Trachemys scripta elegans*, common musk *Sternotherus odoratus* and common snapping turtles *Chelydra serpentina*) were caught frequently enough to assess differences in mortality by bait type.

(1) McKee R.K., Cecala K.K. & Dorcas M.E. (2016) Behavioural interactions of diamondback terrapins with crab pots demonstrate that bycatch reduction devices reduce entrapment. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 26, 1081–1089.
(2) Long J.M., Stewart D.R., Shiflet J., Balsman D., Shoup D.E. (2017) Bait type influences on catch and bycatch in tandem hoop nets set in reservoirs. *Fisheries Research*, 186, 102–108.

Snakes & lizards

- We found no studies that evaluated the effects of using a different bait type on snake and lizard populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Crocodilians

- We found no studies that evaluated the effects of using a different bait type on crocodilian populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

6.38. Change hook baiting technique

- **One study** evaluated the effects of changing the hook baiting technique on reptile populations. This study was in the USA[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One study in the USA[1] found that captive loggerhead turtles were more likely to attempt to swallow thread-baited than single-baited hooks.

Background

How bait is connected to a hook may affect the likelihood of reptiles swallowing the bait and hook together. Changing the baiting technique may involve using whole bait instead of smaller chunks or changing the way that the bait is threaded onto a hook. Single baiting involves putting a fishing hook once through the bait item, whereas thread baiting involves threading the hook through the bait multiple times. As a result, it may be easier for reptiles to strip bait off fishing gear without swallowing the hooks when single baiting is used. Similarly, whole bait rather than chunks of bait may be more easily removed by reptiles.

See also: *Use dyed bait* and *Use a different bait type*.

A study in 2004–2005 in laboratory conditions in Texas, USA (1) found that loggerhead turtles *Caretta caretta* were more likely to attempt to swallow thread-baited than single-baited hooks. The odds that loggerhead turtles would attempt to swallow thread-baited hooks were 2.5 times higher than the odds that they would attempt to swallow single-baited hooks, regardless of bait type used or hook size (results presented as model outputs, see paper for details). Turtle responses to individual baited hooks suspended in their tanks were video recorded (each hook presentation = 1 trial). Sixty 45 cm long captive-reared turtles participated in trials, of which 30 turtles participated again when they reached 55 cm and 65 cm long. Trials were carried out in April and October 2004, and May 2005. Modified circle hooks of different sizes (see paper for details) were baited with whole squid *Illex illecebrosus* or sardines *Sardinella aurita* and either single-baited or thread-baited.

(1) Stokes L.W., Hataway D., Epperly S.P., Shah A.K., Bergmann C.E., Watson J.W. & Higgins B.M. (2011) Hook ingestion rates in loggerhead sea turtles *Caretta caretta* as a function of animal size, hook size, and bait. *Endangered Species Research*, 14, 1–11.

Stakeholder engagement and behaviour change

6.39. Involve fishers in designing and trialling new fishing gear types to encourage uptake of gear that reduces unwanted catch of reptiles

- We found no studies that evaluated the effects on reptile populations of involving fishers in designing and trialling new fishing gear types to encourage uptake of gear that reduces unwanted catch of reptiles.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Involving fishers in designing and trialling new fishing gear types that reduce the unwanted catch ('bycatch') of aquatic reptiles may lead to greater uptake of new geartypes.

See also: *Finance low interest loans to convert to fishing gear that reduces unwanted catch of reptiles* and *Introduce fishing gear exchange programmes to encourage fishers to use gear that reduces unwanted catch of reptiles*.

6.40. Finance low interest loans to convert to fishing gear that reduces unwanted catch of reptiles

- We found no studies that evaluated the effects on reptile populations of financing low interest loans to convert to fishing gear that reduces unwanted catch of reptiles.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Providing financial assistance, such as low interest loans, may encourage fishers to convert to fishing gear types that reduce the unwanted catch ('bycatch') of aquatic reptiles.

See also: *Introduce fishing gear exchange programmes to encourage fishers to use gear that reduces unwanted catch of reptiles* and *Involve fishers in designing and trialling new fishing gear types to encourage uptake of gear that reduces unwanted catch of reptiles*.

6.41. Introduce fishing gear exchange programs to encourage fishers to use gear that reduces unwanted catch of reptiles

- We found no studies that evaluated the effects on reptile populations of introducing fishing gear exchange programs to encourage fishers to use gear that reduces unwanted catch of reptiles.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Introducing fishing gear exchange programmes may encourage fishers to use gear types that reduce the unwanted catch ('bycatch') of aquatic reptiles. Fishers may be provided with alternative gear types that are less harmful to reptiles after surrendering their existing gear. Training on the use of new fishing gear may also be required.

See also: *Involve fishers in designing and trialling new fishing gear types to encourage uptake of gear that reduces unwanted catch of reptiles* and *Finance low interest loans to convert to fishing gear that reduces unwanted catch of reptiles*.

Reduce mortality following unwanted catch

6.42. Establish handling and release procedures for accidentally captured or entangled ('bycatch') reptiles

- **One study** evaluated the effects on reptiles of establishing handling and release procedures for accidentally captured or entangled reptiles. This study was in Canada[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Condition (1 study):** One replicated, controlled study in Canada[1] in a captive setting found that recovery of painted turtles after a long period of being held underwater was similar when turtles recovered out of the water or in the water.

BEHAVIOUR (0 STUDIES)

OTHER (0 STUDIES)

Background

Establishing and implementing best practice protocols for handling and releasing aquatic reptiles entangled or captured in fishing gear may reduce the risk of injury and improve post-release survival. This may involve releasing reptiles with or without delay, using appropriate techniques to remove fishing gear from entangled or hooked reptiles, and using appropriate procedures to release reptiles caught in nets.

For studies on the effect of releasing rehabilitated or accidentally captured ('bycatch') reptiles see *Species management – Rehabilitate and release injured or accidentally caught individuals*.

A replicated, controlled study in 2011 in a laboratory in Ontario, Canada (1) found that short-term recovery of painted turtles *Chrysemys picta* from a lack of oxygen was similar out of water and in water. Recovery from a lack of oxygen was similar for turtles that recovered out of water and those that recovered in water as measured by blood lactate (out of water: 18 mmol/l; in water: 18 mmol/l) and pH (out of water: 7.6; in water: 7.7). Out of water recovery resulted in lower reflex impairment compared to immediately after submergence, whereas in water recovery resulted in similar impairment to both out of water recovery and immediately after submergence (reported as impairment index). Wild-caught male turtles were individually submerged in tanks for 12 hours (held with a cage). Blood lactate, blood pH and reflex response were measured immediately after submergence (6 turtles); after 1 h recovery out of water (7 turtles); after 1 h recovery in water (7 turtles). Reflex response included measuring orientation, startle response, escape response and physical response (see paper for details).

(1) LeDain M.R., Larocque S.M., Stoot L.J., Cairns N.A., Blouin-Demers G. & Cooke S.J. (2013) Assisted recovery following prolonged submergence in fishing nets can be beneficial to turtles: an assessment with blood physiology and reflex impairment. *Chelonian Conservation and Biology*, 12, 172–177.

6.43. Modify fishing gear to reduce reptile mortality in the event of unwanted catch

- **One study** evaluated the effects on reptile populations of using modified gear to reduce reptile mortality in the event of unwanted catch. This study was in the USA[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Survival (1 study):** One replicated study in the USA[1] found that few diamondback terrapins died in crab pots fitted with mesh chimneys.

BEHAVIOUR (0 STUDIES)

Background

Reptiles are vulnerable to drowning when caught in fishing gear and unable to escape. Modifying gear to allow reptiles to reach the surface may keep reptiles alive should they be caught incidentally in fishing gear.

For studies describing the effects of modifying the depth at which fishing gear is deployed, see *Deploy fishing gear at different depths*.

A replicated study in 2012–2013 in five estuarine sites in North Carolina, USA (1) found that pots fitted with a wire mesh chimney led to low mortality of diamondback terrapins *Malaclemys terrapin*. One of 14 terrapins that were caught in pots modified with the chimney died. Standard commercial crab pots (61 cm × 61 cm × 61 cm) were fitted with a chicken wire chimney (122 × 30.5 cm diameter) to provide trapped terrapins with access to air. Pots were deployed June–July in 2012 (4 sites) and 2013 (2 sites). Pots were baited and submerged for 48 hours at a time.

(1) Chavez S. & Williard A.S. (2017) The effects of bycatch reduction devices on diamondback terrapin and blue crab catch in the North Carolina commercial crab fishery. *Fisheries Research,* 186, 94–101.

6.44. Release accidentally caught ('bycatch') reptiles

- **Three studies** evaluated the effects on reptile populations of releasing accidentally caught reptiles. One study was in each of the Caribbean Sea[1], Costa Rica[2] and the Republic of Korea[3].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Survival (2 studies):** One replicated study in the Caribbean Sea[1] found that from a released group of green turtles that included some accidentally caught and some head-started individuals, some survived for at least several months in the wild. One replicated study in the Republic of Korea[3] found that green turtles caught in pound nets all survived for at least two weeks to a year after release.

BEHAVIOUR (1 STUDY)

- **Behaviour change (1 study):** One controlled study off the Pacific coast of Costa Rica[2] found that the behaviour of longline-caught sea turtles following release was broadly similar to free-swimming turtles.

Background

After reptiles have become caught in nets or traps, or hooked on lines, it may be possible to rehabilitate and release individuals depending on the extent or severity of any injuries incurred. Ideally rehabilitation and releases would follow established best practice guidelines to maximise survival rates and ensure each individual is sufficiently healthy prior to release.

For studies that test different methods for safely releasing accidentally captured reptiles, see also *Establish handling and release procedures for accidentally captured or entangled ('bycatch') reptiles*. For studies where injured reptiles were rehabilitated prior to release, see *Species management – Rehabilitate and release injured or accidentally caught individuals*.

A replicated study in 1967–1974 in pelagic waters in the Caribbean Sea near Bermuda (1) found that some accidentally-caught immature and some head-started green turtles *Chelonia mydas* survived at least several months in the wild. In total, 16 of 108 released accidentally-caught or head-started immature green turtles were recaptured. Nine turtles were recovered within 10 months, other recaptured turtles had spent up to 27 months in the wild. Most turtles were recaptured a few hundred metres to 14 km away from their point of release, except for one head-started turtle that was recaptured 2,315 km away from the release site after 10 months. In 1967–1971, eighty-nine green turtles were head-started in Costa Rica and released after approximately two years on the north and south coasts of Bermuda. In addition, 19 wild-born immature green turtles caught accidentally by local fisherman were tagged and released as part of the same programme.

A controlled study in 2001–2003 in pelagic waters on the Pacific coast of Costa Rica (2) found that sea turtles released back into the water after becoming caught on longline hooks travelled similar distances and dived to similar depths compared to free-swimming turtles. Longline-caught turtles travelled similar total distances (117–520 nautical miles) and distances each day (5 nautical miles/day) compared to free-swimming turtles (total distance: 50–443 nautical miles; 5 nautical miles/day). Longline-caught turtles made similar depth daily maximum dives (81–408 m) compared to free-swimming turtles (84–264 m). Longline-caught turtles made more deeper daytime dives than free-swimming turtles, which were more likely to make deeper dives at night (no statistical tests carried out, see original paper for details). None of the longline-caught turtles died during the study; one free-swimming turtle mortality occurred. In total nine olive ridley turtles *Lepidochelys olivacea* and one green turtle *Chelonia mydas* caught by fishermen were monitored using radio tags in November 2001–August 2003. Hooks were removed from the turtle's jaw or mouth (except for one individual, see paper for details) and turtles were released. At the same time, five free-swimming olive ridley turtles were collected by the boat, radio-tagged and released for behavioural comparisons. On average, tags remained on line-caught turtles and free-swimming turtles for 54 and 60 days respectively.

A replicated study in 2015–2017 in three coastal sites of Jeju Island, the Republic of Korea (3) found that after releasing green turtles *Chelonia mydas* that were caught in pound nets near the capture site, all eight turtles survived. Turtles survived for at least 17–314 days and moved 36–1,393 km after being released. Five turtles stayed close to Jeju island and moved 36–581 km; one travelled west toward China and moved 514 km and two travelled east towards Japan and moved 489 and 1,393 km. In August 2015–September 2016, eight healthy turtles (7 juveniles, 1 adult) that were caught accidentally in pound nets (approximate size 25 x 15 x 10 m) were fitted with satellite transmitters before being released near the capture site. Transmitters were attached to the carapace using polyester resin and fiberglass cloth. Turtles were tracked for 17–314 days, with a maximum of one location/individual/day retained for analysis.

(1) Burnett-Herkes J. (1974) Returns of green sea turtles (*Chelonia mydas* Linnaeus) tagged at Bermuda. *Biological Conservation*, 6, 307–308.

(2) Swimmer Y., Arauz R., McCracken M., McNaughton L., Ballestero J., Musyl M., Bigelow K. & Brill R. (2006) Diving behavior and delayed mortality of olive ridley sea turtles *Lepidochelys olivacea* after their release from longline fishing gear. *Marine Ecology Progress Series*, 323, 253–261.

(3) Jang S., Balazs G.H., Parker D.M., Kim B.Y., Kim M.Y., Ng C.K.Y. & Kim T.W. (2018) Movements of Green Turtles (*Chelonia mydas*) Rescued from Pound Nets Near Jeju Island, Republic of Korea. *Chelonian Conservation and Biology*, 17, 236–244.

Logging and wood harvesting

6.45. Thin trees within forests

- **Six studies** evaluated the effects of thinning trees within forests on reptile populations. Three studies were in the USA[2-4] and one was in each of Brazil[1], Spain[5] and Australia[6].

COMMUNITY RESPONSE (3 STUDIES)

- **Richness/diversity (3 studies):** Two replicated, controlled studies (including one randomized study) in the USA[3] and Spain[5] found that areas of thinned forest had similar reptile species richness compared to areas with no thinning. One study[5] also found that thinned areas had lower species richness than areas of open habitat. One replicated, controlled study in Australia[6] found that areas of forest thinned 8–20 years previously had higher diversity of reptiles than areas thinned less than eight or more than 20 years previously, or than areas with no thinning.

POPULATION RESPONSE (6 STUDIES)

- **Abundance (6 studies):** Two of four replicated, controlled studies (including two randomized studies) in Brazil[1], the USA[3,4], and Spain[5] found that areas of thinned forest had a similar abundance of reptiles compared to areas with no thinning[3,4]. One study[1] found mixed effects of thinning trees on the abundance of three lizard species. The other study[5] found that areas of thinned forest had a higher abundance of reptiles than areas with no thinning. That study[5] also found that areas with the most thinning had a similar abundance of reptiles compared to areas of open habitat. One replicated, controlled study in Australia[6] found that areas of forest thinned 8–20 years previously had a higher abundance of reptiles than areas thinned at other times or areas with no thinning. One replicated, randomized, controlled study in the USA[2] found that areas of thinned forest had a higher abundance of snakes than clearcut forest.

BEHAVIOUR (0 STUDIES)

Background

Thinning is a forestry practice where selective removal of trees is used to increase the size and health of remaining trees in both wild-harvested and plantation forests. Ecological thinning (pre-commercial thinning) is a variant of thinning used in forest conservation where the primary aim is to increase growth of trees, but the secondary aim is to develop or improve wildlife habitat (e.g. hollows, sun gaps).

For studies of the effect of prescribed fires on their own and in combination with vegetation cutting, see *Threat: Natural system modifications – Use prescribed burning* and *Use prescribed burning in combination with vegetation cutting*; *Use prescribed burning in combination with herbicide application* and *Use prescribed burning in combination with grazing*.

A replicated, randomized, controlled study in 1987–1997 in tropical forest in Amazonas, Brazil (1) found that after killing ('girdling') non-commercial tree species of >25 cm trunk diameter, density of one of three lizard species was reduced 12 years later compared to unmanaged areas. Twelve years after non-commercial trees were girdled, striped whiptail lizard *Kentropyx calcarata* density (1–3 lizards/plot) was reduced compared to in unmanaged areas (4–6 lizards/plot), but density was similar for giant ameiva *Ameiva ameiva* (girdled: 1–4 individuals/plot; no management: 0–5 individuals/plot) and black-spotted skink *Mabuya nigropunctata* (girdled: 0–1 individuals/plot; no management: 0–2 individuals/plot). In 1985, non-commercial trees >25 cm diameter at breast height were killed ('girdled', see original paper for details) in three forest plots (4 ha each). Lizards were surveyed in girdled plots and three 4 ha plots with no historical management on foot by walking six 200 x 20 m transects in each plot during daytime in August–October 1996 and July 1997. The maximum number of lizards counted/plot was used as a measure of density.

A replicated, randomized, controlled study in 2004–2006 in pine forests in South Carolina, USA (2) found that snake abundance was higher in thinned compared to clearcut forest. The number of snakes captured was higher after thinning (180 individuals) compared to clearcutting (80–102 individuals). Numbers of snakes captured in unharvested plots was 137. Four circular forest sites were divided into four plots and each plot was randomly assigned one of four treatments: 85% thinned, clearcut with coarse woody debris retained, clearcut with coarse woody debris removed and unharvested for >30 years. Logging was from February to April 2004. Reptiles were sampled using drift fences with pitfall traps. Traps were checked every 1–2 days from April 2004 to July 2006 except for August.

A replicated, randomized, controlled study in 2006–2007 in hardwood forests in North Carolina, USA (3) found that overall reptile species richness and capture rates were similar in areas with tree thinning compared to unmanaged areas. Overall reptile richness and overall reptile, snake and turtle captures were similar after thinning by mechanical cutting (richness: 6–7 species/100 array nights, overall captures: 6 individuals/100 array nights, snakes: 1–2 individuals/100 array nights, turtles: 0 individuals/100 array nights) and no management (richness:6, overall captures: 7, snakes: 3–5, turtles: 0). Three plots each (10 ha) were managed with mechanical-cutting (using chainsaws to cut trees and understory, 2001–2002) or not managed. Reptiles were surveyed in May–August 2006 and 2007 using drift fences with pitfall traps ('arrays', 3/site).

A replicated, controlled, before-and-after study in 2005–2008 in mixed forest in Alabama, USA (4) found no clear effects of thinning on the abundance of six reptile species when compared to areas that were left unmanaged. The abundance of all six species (eastern fence lizard *Sceloporus undulatus*, green anole *Anolis carolinensis*, little brown skink *Scincella lateralis*, five-lined skink *Plestiodon fasciatus*, copperhead *Agkistrodon contortrix* and eastern racer *Coluber constrictor*) remained similar following heavy and light thinning compared to unmanaged areas (see paper for individual species abundances). In 2005–2008, three 9 ha plots each were either lightly thinned (17 m^2/ha tree retention), heavily thinned (11 m^2/ha tree retention) or left unmanaged (9 plots in total). Reptiles were surveyed for 3–6 months before management began (564 total trap nights in April–August) and in the two years after management (3,132 total trap nights in March–September) using drift fences with pitfall traps. Individuals were marked before release.

A replicated, controlled study in 2014 in pine forest in Granada, Spain (5) found that thinning trees in commercial forest by 66% increased reptile abundance but not species richness compared to thinning by 50% or no thinning. Reptile abundance was greater in 66%-thinned forest (11 reptiles/plot) compared to 50%-thinned (3 reptiles/plot) or unthinned forest (3 reptiles/plot) but similar to reptile abundance in open landscape (9 reptiles/plot). Reptile species richness was similar in 66%-thinned (2 species/plot), 50%-thinned (1 species/plot) and unthinned forest (1 species/plot), but lower than species richness in open landscape (3 species/plot). In 2010, a pine plantation with 600 trees/ha was managed by thinning 66% and 50% of trees in 20–37 ha areas. Reptiles were surveyed using a visual encounter method along u-shaped line transects in May-June 2014 in four plots each of 66% thinning, 50% thinning, as well as in four plots each with no tree thinning and in adjacent open landscape (all plots 100 x 35 m, 16 total plots). Each plot was surveyed four times at least five days apart.

A replicated, controlled study in 2015–2016 in pine and eucalypt woodland in north-west New South Wales, Australia (6) found that reptile abundance and diversity were higher 8–20 years after tree thinning compared to <8 years or >20 years after thinning. Reptile abundance was 2–4.5 times greater and reptile diversity was 1.4–1.5 times greater 8–20 years after thinning than in unthinned, thinned <8 years ago, thinned >20 years ago or undisturbed forest (data reported as statistical model outputs, see original paper for details). In total 85 reptiles of 21 different species were caught across all sites (see original paper for changes in individual species abundances). The effect of tree thinning on reptiles was monitored in five 20–30 ha plots in 30 historically-managed forestry sites (non-commercial and commercial). Plots had the following thinning history: thinned <8 years ago using mechanical and manual brushcutting (thinnings left on site), thinned 8–20 years ago (larger stems for saw logs were removed from the site), thinned >20 years ago (thinnings left on site), unthinned (~6,500 stems/ha) and long undisturbed (see original paper for details). Reptiles were surveyed along a 200 m transect in each plot using nocturnal spotlighting (sampled once/plot, dates not provided) and drift fence/pitfall trap arrays (two traps/array, two arrays/plot) in October–November 2015 (eight days) and March 2016 (four days).

(1) Lima A.P., Suarez F.I.O. & Higuchi N. (2001) The effects of selective logging on the lizards *Kentropyx calcarata*, *Ameiva ameiva* and *Mabuya nigropunctata. Amphibia-Reptilia*, 22, 209–216.

(2) Todd B.D. & Andrews K.M. (2008) Response of a reptile guild to forest harvesting. *Conservation Biology*, 22, 753–761.

(3) Matthews C.E., Moorman C.E., Greenberg C.H. & Waldrop, T.A. (2010) Response of reptiles and amphibians to repeated fuel reduction treatments. *The Journal of Wildlife Management*, 74, 1301–1310.

(4) Sutton W.B., Wang Y. & Schweitzer C.J. (2013) Amphibian and reptile responses to thinning and prescribed burning in mixed pine-hardwood forests of northwestern Alabama, USA. *Forest Ecology and Management*, 295, 213–227.

(5) Azor J.S., Santos X. & Pleguezuelos J.M. (2015) Conifer-plantation thinning restores reptile biodiversity in Mediterranean landscapes. *Forest Ecology and Management*, 354, 185–189.

(6) Gonsalves L., Law B., Brassil T., Waters C., Toole I. & Tap P. (2018) Ecological outcomes for multiple taxa from silvicultural thinning of regrowth forest. *Forest Ecology and Management*, 425, 177–188.

6.46. Coppice trees

- **One study** evaluated the effects of coppicing trees on reptile populations. This study was in the UK[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Abundance (1 study):** One replicated, site comparison study in the UK[1] found that no slow worms or common lizards were found in coppiced areas of woodland, whereas they were found in open areas maintained by vegetation cutting.

BEHAVIOUR (0 STUDIES)

Background

Coppicing is a management practice typical of Eurasian northern temperate zone deciduous woodlands and wood pastures, in which stems of tree species, such as hazel *Corylus avellana* and sweet chestnut *Castanea sativa*, are cut near ground level once every few years, often in defined coppice compartments. These then regrow from the cut 'stool' giving a sustainable yield of woody material harvested on a rotational basis. Coppicing maintains a mosaic of woodland areas with differing amounts of daylight reaching the forest floor and, therefore, promotes a variety of ground vegetation conditions. This may benefit reptiles that require either open canopy woodland or a mix of open and more closed woodland in close proximity. Coppicing has declined over the last century and some former coppice woodlands are no longer actively managed.

A replicated, site comparison study (year not provided) in two sites of temperate broadleaf woodland on the border of Northamptonshire and Cambridgeshire, UK (1) found that in coppiced areas of a woodland no slow worms *Anguis fragilis* or common lizards *Zootoca vivipara* were found, whereas both species were found in open areas maintained by cutting. No slow worms or common lizards were found in either recently coppiced sites (2–6 years previously) or older coppiced sites (9–17 years old), whereas 41 common lizards and 102 slow worms were found in open areas maintained by cutting. In each of two areas of woodland, three sites of recently coppiced woodland (2–6 years old), three sites of older coppice (9–17 years old) and three open areas were selected (one of the open areas was selected two weeks after surveys began). All coppiced areas were dominated by small-leaved lime trees *Tilia cordata*. At each survey site, 20 coverboards (50 x 50 cm; 10 made of roofing felt, 10 made of corrugated bitumen) were arranged in a grid, with 5 m gaps between boards. Coverboards were left for one week, and then checked for reptiles on 3–6 days/week for eight weeks.

(1) Fish A.C.M. (2015) Common lizards (*Zootoca vivipara*) and slow-worms (*Anguis fragilis*) are not found in coppiced Small-Leaved Lime (*Tilia cordata*) areas of a Northamptonshire-Cambridgeshire Nature Reserve. *Herpetological Bulletin*, 134, 26–27.

6.47. Retain riparian buffer strips during timber harvest

- We found no studies that evaluated the effects of retaining riparian buffer strips during timber harvest on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Retaining forest strips along water courses or around ponds during timber harvest can help mitigate the effects of habitat loss and disturbance for forest species. They can also help sustain the microclimate and reduce potential problems such as soil erosion. Retained habitat strips also provide corridors for dispersal.

For other studies on the effects of buffer strips see *Threat: Agriculture – Create uncultivated margins around arable or pasture fields; Threat: Pollution – Plant riparian buffer strips;* and *Habitat protection – Retain buffer zones around core habitat.*

6.48. Leave standing/deadwood snags in forests

- **Two studies** evaluated the effects of leaving standing/deadwood snags in forests on reptile populations. Both studies were in the USA[1,2].

COMMUNITY RESPONSE (2 STUDIES)

- **Richness/diversity (2 studies):** One of two replicated, randomized, controlled studies in the USA[1,2] found that adding snags and woody debris had mixed effects on reptile diversity and species richness when compared to not manipulating debris or removing debris[1]. The other study[2] found that increasing standing coarse woody debris had no effect on reptile diversity and species richness.

POPULATION RESPONSE (2 STUDIES)

- **Abundance (2 studies):** One of two replicated, randomized, controlled studies in the USA[1,2] found that adding snags and woody debris had mixed effects on reptile abundance when compared to not manipulating debris or removing debris[1]. The other study[2] found that increasing standing coarse woody debris had no effect on reptile abundance.

BEHAVIOUR (0 STUDIES)

Background

Standing and prone deadwood provides important microhabitat for some species and so removal of dead vegetation, for example to manage fire risk, may limit the distribution, abundance and species richness of reptiles in local areas (James & M'Closkey 2003).

For studies discussing leaving woody debris in place after logging or wood harvesting, see *Leave woody debris in forests after logging*. For studies discussing adding woody debris back to landscapes, see *Habitat restoration and creation – Add woody debris to landscapes.*

James S.E. & M'Closkey R.T. (2003) Lizard microhabitat and fire fuel management. *Biological Conservation*, 114, 229–293.

A replicated, randomized, controlled study in 1998–2005 of pine stands in South Carolina, USA (1, same experimental set-up as 2) found that the creation or removal of snags (standing dead trees) had no effect on reptile abundance, species richness and diversity compared to not manipulating debris in forests. In two trials, reptile abundance, species richness and diversity was similar between plots with snags added (abundance: 0.2–0.3 individuals/plot, richness: 5–6 species, diversity: 10–13 Shannon-Weiner index), or all snags and coarse woody debris removed (0.3–0.5, 6–7, 13–17), compared to not manipulating debris (0.3–0.4, 7, 13–17). In the second trial, reptile abundance, richness and diversity were lower when standing snags were added (0.3 individuals/plot, 5 species, 10 Shannon-Weiner index, respectively) compared to when all woody debris was removed (0.5, 7, 17). Snake abundance was higher with woody debris removal compared to snag addition (debris removal: 0.2 individuals/plot; snags added: 0.1), but lizard abundance was not (debris removal: 0.3 individuals/plot; snags added: 0.2). Treatments were randomly assigned to 9 ha plots within three forest blocks in 1996–2001: standing snag addition (10 fold increase), all woody debris removal, downed woody debris addition (five-fold increase), and no manipulation and in 2002–2005: downed woody debris addition, woody debris removal, standing snag addition, and no manipulation. Reptiles were sampled using drift fences with pitfall traps in 1998–2005.

A replicated, randomized, controlled study in 1996–2008 in a loblolly pine *Pinus taeda* forest in South Carolina, USA (2, same experimental set-up as 1) found that increasing standing coarse woody debris had no effect on reptile abundance, species richness or diversity. Abundance, species richness and diversity were similar between plots with increased standing woody debris (abundance: 0.18

individuals/m fencing, richness: 0.10 species/m fencing, diversity: 0.03 Shannon-Wiener Index) and plots with no manipulation of debris (0.15, 0.11, 0.03). Nine-ha plots within three pine stands (approximately 45 years old) were randomly assigned the following management: standing woody debris increased 10 fold by girdling then injecting with herbicide (initiated 2001, to 35 m^3/ha woody debris in 2007) or no manipulation of woody debris (initiated 1996, 13 m^3/ha woody debris). All plots were prescribed burned in 2004. Reptiles were sampled for 14 days/plot in each of seven seasons (January 2007–August 2008) using drift fences with pitfall traps.

(1) Owens A.K., Moseley K.R., McCay T.S., Castleberry S.B., Kilgo J.C. & Ford W.M. (2008) Amphibian and reptile community response to coarse woody debris manipulations in upland loblolly pine (*Pinus taeda*) forests. *Forest Ecology and Management*, 256, 2078–2083.

(2) Davis J.C., Castleberry S.B. & Kilgo J.C. (2010) Influence of coarse woody debris on herpetofaunal communities in upland pine stands of the southeastern Coastal Plain. *Forest Ecology and Management*, 259, 1111–1117.

6.49. Leave woody debris in forests after logging

- **Six studies** evaluated the effects of leaving woody debris in forests after logging on reptile populations. All six studies were in the USA[1-6].

COMMUNITY RESPONSE (5 STUDIES)

- **Richness/diversity (5 studies):** Four of five studies (including four replicated, randomized, controlled studies) in the USA[1-3,5,6] found that leaving or removing woody debris did not affect the richness of reptile species[3,5,6], or immigrating reptiles[2]. The other study[1] found that areas where woody debris was left in place had higher reptile species richness than areas where debris was cleared and burned. Three replicated, randomized, controlled studies in the USA[3,5,6] found that leaving or removing woody debris did not affect reptile species diversity[3,5] or overall reptile and amphibian species diversity[6].

POPULATION RESPONSE (5 STUDIES)

- **Abundance (5 studies):** Four of five studies (including three replicated, randomized, controlled studies) in the USA[1-5] found that leaving or removing woody debris did not affect the abundance of reptiles[3], snakes[4], snakes and lizards[5] or immigrating reptiles[2]. The other study[1] found that areas where woody debris was left in place had higher reptile abundance than areas where debris was cleared and burned.

BEHAVIOUR (0 STUDIES)

Background

Retaining woody debris after logging preserves some shelter habitat for reptiles and may mitigate some of the impacts of logging.

For studies discussing leaving deadwood snags in place after logging or wood harvesting, see *Leave standing/deadwood snags in forests*. For studies discussing adding woody debris back to landscapes, see *Habitat restoration and creation – Add woody debris to landscapes.*

A site comparison study in 1978–1982 in pine forests in Florida, USA (1) found that when woody debris was retained following clearcutting prior to replanting, reptile species richness and abundance was higher than when ground cover was cleared and burned prior to replanting. Three years after woody debris was retained prior to replanting, reptile species richness and abundance were higher (richness: 13 species/trapping array, abundance: 41 individuals/array) than after ground cover was cleared and burned (richness: 9, abundance: 19) and similar to uncut forest (richness: 15, abundance: 52). Overall burrow and refugia-dwelling reptiles and amphibians were more abundant in areas where woody debris was retained compared to where ground cover was cleared and burned or in uncut forest (see paper for individual species abundances). Two of three sites (49–140 ha) were clearcut in 1978 and then managed by retaining woody debris cover (in 1978: 59% harvested by chainsaw, January–August 1979: roller chopped twice) or clearing and burning cover (1978: 74% harvested by feller-buncher, January–August 1979: stump removal, burned, harrowed) prior to replanting in September–November 1979. Reptiles were sampled weekly from August 1981 to October 1982 using four drift fence arrays (four 7.5 x 50 cm galvanised flashing fences in a plus-shape with three aluminium window screen funnel traps on each arm) at all three sites.

A replicated, randomized, controlled study in 1997–1999 in forest and wetlands in South Carolina, USA (2) found that overall abundance and richness of reptiles immigrating to wetlands were similar after clearcutting with cut debris left in place or after clearcutting with replanting compared to before management. After six and 18 months, overall richness and abundance of immigrating reptiles were statistically similar between clearcutting with debris left in place (average change in richness: 45–66% decline, abundance: 54–79% decline), clearcutting with replanting (27–59% decline, 43–70% decline) and no harvesting (28–72% decline, 51–77% decline) compared to before management was carried out. See original paper for details of groups of and individual species changes in immigration compared to before management. Pine *Pinus spalustris* plantations (<10 ha each) surrounding five wetlands (0.4–1.1 ha) were divided into three and managed in June 1998 by: clearcutting with residual woody debris/slash left in place, clearcutting with replanting (including mechanical site preparation prior to planting), or no harvesting. Reptile movements from the adjacent wetlands were monitored by enclosing each wetland with a drift fence, with pairs of pitfall traps placed every 10 m along the fence. Pitfalls were checked daily in June–December 1997 (pre-treatment), 1998 (6 months post-treatment) and 1999 (18 months post-treatment). Captured individuals were individually marked using toe clipping, PIT tags or shell notching.

A replicated, randomized, controlled study in 1998–2005 of pine stands in South Carolina, USA (3, same experimental set-up as 5) found that leaving coarse woody debris in place had no effect on reptile abundance, species richness and diversity compared to removing it. In two trials, removing all downed and standing woody debris did not change reptile richness (debris removed: 5–7 species), diversity (10–17, Shannon-Weiner index) and abundance (0.2–0.5 individuals/plot/night) compared to not manipulating woody debris (richness: 7, diversity: 13–17, abundance: 0.3–0.4). The two treatments were randomly assigned to 9 ha plots within three forest blocks in 1996–2001 and 2002–2005: all woody debris removal or no manipulation. Five drift-fence arrays with pitfall traps/plot were used for sampling in 1998–2005.

A replicated, randomized, controlled study in 2004–2006 in pine forests in South Carolina, USA (4) found that snake abundance was similar in clearcut forest with woody debris left in place compared to when debris was removed, but lower in clearcut compared to thinned forest. The number of snakes captured was similar after clearcutting and leaving coarse woody debris in place (102 individuals) or removing coarse woody debris (80 individuals), but lower than after thinning (180 individuals). Numbers of snakes captured in unharvested plots was 137. Four circular forest sites were divided into four plots and each plot was randomly assigned one of four treatments: clearcut with coarse woody debris retained, clearcut with coarse woody debris removed, 85% thinned and unharvested for >30 years. Logging was from February to April 2004. Reptiles were sampled using drift fences with pitfall traps. Traps were checked every 1–2 days from April 2004 to July 2006 except for August.

A replicated, randomized, controlled study in 1996–2008 in a loblolly pine *Pinus taeda* forest in South Carolina, USA (5, same experimental set-up as 3) found that leaving downed coarse woody debris had no effect on lizard or snake abundance, species richness or diversity compared to removing debris. After retaining woody debris, snake abundance, richness and diversity were similar (abundance: 0.04 individuals/m drift fence, richness: 0.04 species/m drift fence, diversity: 0.01 Shannon-Wiener Index) compared to when debris was removed (abundance: 0.07, richness: 0.04, diversity: 0.01) and also similar to when debris was added (abundance: 0.03, richness: 0.02, diversity: 0.003). For lizards there was also no difference between retaining (abundance: 0.01, richness: 0.07, diversity: 0.02), removing (abundance: 0.15, richness: 0.07, diversity: 0.02) or adding debris (abundance: 0.15, richness: 0.07, diversity: 0.02). Nine ha plots in three pine stands (approximately 45 years old, three plots/stand) were managed by: retaining woody debris (initiated 1996, 13 m^3/ha woody debris); removing all downed woody debris ≥10 cm diameter and ≥60 cm in length by hand (initiated 1996, to 0.24 m^3/ha in 2006); or increasing volume of downed woody debris five-fold by felling trees (initiated 2001, to 59 m^3/ha in 2007). All plots were prescribed burned in 2004. Reptiles were sampled for 14 days/plot in each of seven seasons (January 2007–August 2008) using drift fences with pitfall traps.

A replicated, randomized, controlled study in 2010–2014 in commercial pine forests in North Carolina and Georgia, USA (6) found that retaining woody debris after clearcutting did not affect reptile species richness, or overall reptile and amphibian species diversity. Over 3–4 years after clearcutting, reptile species richness and overall reptile and amphibian species diversity were similar when 100% of woody debris was retained, 15–30% of wood debris was retained, or all debris was removed (results reported as statistical model outputs, see original paper for details). Eight replicate sites in three locations (one in North Carolina, two in Georgia) of intensively managed loblolly pine *Pinus taeda* plantations (64–70 ha/site, 25–35 years old) were clearcut in autumn 2010–summer 2011 and six 11–12 ha plots/site were managed by retaining 100% of woody debris; retaining 30% of woody debris in large piles; retaining 30% of woody debris evenly distributed; retaining 15% of woody debris in large piles; retaining 15% of woody debris evenly distributed; or by removing all woody debris (following traditional practice). Sites were replanted and treated with herbicide in 2011–2012. Reptiles and amphibians were surveyed in April–August 2011–2014 in North Carolina and

2011–2013 in Georgia using three drift fence and funnel trap arrays/plot. Three–eight trapping periods were carried out/year (2011: 10 consecutive days; 2012–2014: five consecutive days).

(1) Enge K.M. & Marion W.R. (1986) Effects of clearcutting and site preparation on herpetofauna of a north Florida flatwoods. *Forest Ecology and Management*, 14, 177–192.

(2) Russell K., Hanlin H., Wigley T. & Guynn D. (2002) Responses of isolated wetland herpetofauna to upland forest management. *The Journal of Wildlife Management*, 66, 603–617.

(3) Owens A.K., Moseley K.R., McCay T.S., Castleberry S.B., Kilgo J.C. & Ford W.M. (2008) Amphibian and reptile community response to coarse woody debris manipulations in upland loblolly pine (*Pinus taeda*) forests. *Forest Ecology and Management*, 256, 2078–2083.

(4) Todd B.D. & Andrews K.M. (2008) Response of a reptile guild to forest harvesting. *Conservation Biology*, 22, 753–761.

(5) Davis J.C., Castleberry S.B. & Kilgo J.C. (2010) Influence of coarse woody debris on herpetofaunal communities in upland pine stands of the southeastern Coastal Plain. *Forest Ecology and Management*, 259, 1111–1117.

(6) Fritts S., Moorman C., Grodsky S., Hazel D., Homyack J., Farrell C. & Castleberry S. (2016) Do biomass Harvesting Guidelines influence herpetofauna following harvests of logging residues for renewable energy? *Ecological Applications*, 26, 926–939.

6.50. Use smaller machinery to log forests

- We found no studies that evaluated the effects of using smaller machinery to log forests on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Logging activities cause disturbances that may have negative impacts on reptiles. Using smaller machinery may reduce the level of disturbance, with benefits to reptile populations.

6.51. Use patch retention harvesting instead of clearcutting

- We found no studies that evaluated the effects of using patch retention harvesting instead of clearcutting on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Removing trees, through clearcutting or clearfelling is likely to have substantial effects on reptiles, through alteration of habitat and removal of food and shelter. Patch retention is the act of leaving groups of trees during harvesting, which may act as refugia to support forest fauna and enable its recolonization of the remainder of the forest as it regrows.

6.52. Harvest groups of trees instead of clearcutting

- We found no studies that evaluated the effects of harvesting groups of trees instead of clearcutting on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Forests naturally undergo disturbances such as storms and lightning that can create open patches. Similarly, harvesting groups of trees rather than clearcutting forest creates a mix of different habitats, allowing a greater range of species to survive in a forest.

6.53. Use shelterwood harvesting

- **Two studies** evaluated the effects of shelterwood harvesting on reptile populations. Both studies were in the USA[1,2].

COMMUNITY RESPONSE (1 STUDY)

- **Richness/diversity (1 study):** One replicated, randomized, controlled, before-and-after study in the USA[2] found that shelterwood harvesting had mixed effects on reptile species richness compared to areas with no management.

POPULATION RESPONSE (2 STUDIES)

- **Abundance (2 studies):** One replicated, randomized study in the USA[1] found that areas with shelterwood harvesting had a lower abundance of juvenile eastern box turtles than clearcut areas. One replicated, randomized, controlled, before-and-after study in the USA[2] found that shelterwood harvesting had mixed effects on reptile abundance compared to areas with no management.

BEHAVIOUR (0 STUDIES)

Background

Shelterwood harvesting is a management technique designed to obtain even-aged forests. It involves harvesting trees in a series of partial cuts, with trees removed uniformly over the plot. This allows new seedlings to grow from the seeds of older trees. This can help to maintain distinctive forest species and increase forest structural diversity.

A replicated, randomized study in 2001–2005 in three sites of secondary broadleaf forest in Alabama, USA (1) found that using shelterwood harvesting resulted in lower abundance of juvenile eastern box turtles *Terrapene carolina Carolina* compared to areas that were clearcut. Abundance was lower in shelterwood plots (0.001 turtles/trap night) compared to clearcut plots (0.002 turtles/trap night). In autumn 2001, three sites were split in to three plots (4 ha plots), and plots were randomly selected for shelterwood harvesting (25–50% tree retention; 2 plots/site) or clearcutting (0 % retention, 1 plot/site). Trees were felled with a chainsaw and dragged out (using a grapple skidder). In July–August 2002 and March–September 2003–2005, three drift fences (15 m long) and three artificial pools for capturing reptiles (91 x 61 x 46 cm, buried in centre of each plot) were installed in each plot. Drift fences were opened intermittently for periods of five days and checked daily for a total of 1,455–1,575 trap nights/patch.

A replicated, randomized, controlled, before-and-after study in 2008–2014 in an upland mixed oak forest in the Appalachians, USA (2) found that shelterwood harvesting increased lizard but not overall reptile and snake species richness and abundance compared to no management. Overall lizard species richness and capture rates increased after shelterwood harvesting (species richness: 0.8–1.5 species/100 fence nights, abundance: 0.5–1.3 individuals/100 fence nights) compared to no management (0, 0–0.1). Overall reptile and snake species richness and abundance were similar after shelterwood harvesting (overall reptile abundance: 0.7–1.7 captures/100 fence nights), compared to no management (overall reptile abundance: 0.2–0.7 captures/100 fence nights; snake abundance and all reptile and snake species richness data presented as model outputs). See paper for changes in individual species abundances. Shelterwood harvesting was carried out in 2009–2010 in 4–5 replicate plots of 225 x 225 m. Trees were felled with chainsaws and grapple cutters and dragged to log landings. Monitoring took place using drift fences, pitfall and funnel traps in May-August one year pre-treatment (2008) and five years post treatment (sampled in 2010, 2011, 2013, 2014). Plots of the same size and number without any management applied were monitored at the same time.

(1) Felix Z., Wang Y., Czech H. & Schweitzer C.J. (2008) Abundance of juvenile eastern box turtles relative to canopy cover in managed forest stands in Alabama. *Chelonian Conservation and Biology*, 7, 128–130.

(2) Greenberg C.H., Moorman C.E., Raybuck A.L., Sundol C., Keyser T.L., Bush J., Simon D.M. & Warburton G.S. (2016) Reptile and amphibian response to oak regeneration treatments in productive southern Appalachian hardwood forest. *Forest Ecology and Management*, 377, 139–149.

6.54. Use selective logging

- **Three studies** evaluated the effects of using selective logging in forests on reptile populations. One study was in each of Brazil[1], the USA[2] and Mexico[3].

COMMUNITY RESPONSE (1 STUDY)

- **Richness/diversity (1 study):** One replicated, site comparison study in Mexico[3] found that areas with low intensity selective logging tended to have similar reptile species richness compared to areas with high intensity selective logging.

POPULATION RESPONSE (2 STUDIES)

- **Abundance (2 studies):** One of two replicated, randomized, controlled studies (including one before-and-after study) in Brazil[1] and the USA[2] found that selective logging intensity had mixed effects on the abundance of three lizard species. The other study[2] found that areas with selective logging had similar reptile abundance compared to areas with combined clearcutting and thinning.

- BEHAVIOUR (0 STUDIES)

Background

Selective logging is a more ecologically sustainable practice than clearcutting, which entails removing all trees at the same time. The idea behind selective logging is to maintain an uneven or all-aged forest of trees varying not only in age, but in size and species as well. Selective logging of mature forest involves targeting the harvest of trees based on specific criteria. This might be based on size of tree, tree health, or tree species, for example.

A replicated, randomized, controlled study in 1987–1997 in tropical forest in Amazonas, Brazil (1) found that greater selective logging intensity increased the density of one of three lizard species, but density of all three lizard species reduced over time after logging. Black-spotted skink *Mabuya nigropunctata* density increased with logging intensity, but striped forest whiptail lizard *Kentropyx calcarata* and giant ameiva *Ameiva ameiva* densities did not (data are presented as statistical model outputs). Black-spotted skink and giant ameiva densities were higher in plots with trees felled 4 years earlier, compared to plots where trees were felled 9–10 years earlier, or in unmanaged plots. Whiptail lizard density was higher in plots felled 4 years earlier than 9–10 years earlier, but density in plots felled 9–10 years earlier was no different in unmanaged plots (see original paper for details). In three blocks (24 ha each), forest plots (4 ha each) were managed as follows: commercial tree felling using selective logging in 1987 (two plots), in 1988 (one plot), in 1993 (one plot), or no management (one plot). The reduction in wood volume after logging ranged from 44–107 m^3/ha (including commercial trees and those accidentally felled/killed by logging operations). Lizards were surveyed on foot by walking six 200 m x 20 m transects in each plot during daytime in August–October 1996 and July 1997. The maximum number of lizards counted/ plot was used as a measure of density.

A replicated, randomized, controlled, before-and-after study in 1992–2000 in oak-pine and oak-hickory forest in Missouri, USA (2) found that there was no difference in reptile abundance between sites with small group or single tree selection harvesting and those with combined clearcutting and thinning, although four of six species of reptiles increased in abundance in clearcut and thinned sites after management began. Overall, abundances of six reptile species were similar between small group or single tree selection harvesting and clearcutting with thinning and unmanaged sites (results reported as statistical model outputs). Abundances of four of six species increased after clearcutting and thinning took place compared to before management (see original paper for details). Nine sites (312–514 ha) were randomly assigned to treatments: small group or single tree selection harvesting (5% area; uneven-aged management), clearcutting in 3–13 ha blocks (10–15% total area) with forest thinning (even-aged), or no management (3 sites/treatment). Harvesting was in May 1996 and 1997. Twelve drift-fence arrays with pitfall and funnel traps were established/site. Traps were checked every 3–5 days in spring and autumn 1992–1995 (before management) and 1997–2000 (after management).

A replicated, site comparison study in 2013–2015 in mixed forest in Oaxaca, Mexico (3) found that reptile diversity but not richness tended to be higher following low intensity selective logging compared to high intensity logging, but lower than in unlogged forest. Results were not statistically tested. Reptile richness was 2–4 species after low intensity and 3–5 species after high intensity logging compared to 13 species in unlogged forest. Reptile diversity was lowest in high intensity logged forest after five years of recovery (Shannon-Wiener Index: 2 effective species) compared to low intensity logged forest after one year of recovery (3) or unlogged forest (7). See paper for details of individual species abundances. Reptiles were monitored monthly in August 2013–July 2015 in two 0.1 ha plots in forest stands at 1, 5 and 10 years after high intensity (elimination of canopy cover, promotes homogeneous forest stands; 6 plots) and lower intensity logging (fewer trees removed as part of 'group selection logging'; 6 plots) and in forest that had not been logged for at least 35 years (2 plots). Reptiles were surveyed using intensive searches (1,344 total man hours) and pitfall traps (64,512 total trap hours) and identified to species level after capture.

(1) Lima A.P., Suarez F.I.O. & Higuchi N. (2001) The effects of selective logging on the lizards *Kentropyx calcarata*, *Ameiva ameiva* and *Mabuya nigropunctata. Amphibia-Reptilia*, 22, 209–216.
(2) Renken R.B., Gram W.K., Fantz D.K., Richter S.C., Miller T.J., Ricke K.B., Russell B. & Wang X. (2004) Effects of forest management on amphibians and reptiles in Missouri Ozark forests. *Conservation Biology*, 18, 174–188.
(3) Aldape-Lopez C.T. & Santos-Moreno A. (2016) Effect of forest management on the herpetofauna of a temperate forest of western Oaxaca, Mexico. *Revista De Biologia Tropical*, 64, 931–943.

6.55. Reseed logged forest

- **One study** evaluated the effects of reseeding logged forest on reptile populations. This study was in the USA[1].

COMMUNITY RESPONSE (1 STUDY)

- **Community composition (1 study):** One replicated, site comparison study in the USA[1] found that reptile communities in areas that were reseeded were not more similar to mature forest stands than those left to regenerate naturally.
- **Richness/diversity (1 study):** One replicated, site comparison study in the USA[1] found that areas that were reseeded had similar reptile species richness and diversity compared to areas left to regenerate naturally.

POPULATION RESPONSE (1 STUDY)

- **Abundance (1 study):** One replicated, site comparison study in the USA[1] found that areas that were reseeded had similar reptile abundance compared to areas left to regenerate naturally.

BEHAVIOUR (0 STUDIES)

Background

This intervention involves re-seeding logged forest to re-establish tree communities after cutting, as opposed to leaving forest to regenerate naturally.

A replicated, site comparison study in 1991–1992 in sand-pine scrub forest in Florida, USA (1) found that clearcutting with reseeding did not have greater reptile species richness, abundance, or diversity than salvage logging with natural regeneration but that community composition differed between managed and unmanaged stands. Logging and seeding treatments were carried out together and it is not possible to distinguish their effects. Reptile species richness, abundance, diversity and evenness were similar between clearcutting with broadcast seeded (richness: 8 species/stand, abundance: 69 individuals/stand, Shannon Diversity Index: 0.7, evenness: 0.8, see paper for details), clearcutting with direct-drill seeded plots (7, 79, 0.6, 0.8), salvaged logged with natural regeneration (9, 41, 0.8, 0.8), and unmanaged stands (9, 31, 0.8, 0.8). Community composition was similar between all managed plots (0.88–0.95), but all managed plots were less similar to mature forest stands (0.54–0.65, results reported as Horn's Index of Community Similarity, see original paper for details including individual species abundances). Forest stands were managed by: clearcutting with roller chopping and broadcast seeding, clearcutting with direct-drill machine-seeding and salvage logging with natural regeneration (following a high intensity wildfire) (three stands/management type). Reptiles were surveyed 5–7 years after management in August 1991–September 1992 by trapping every alternate two weeks using drift fences with pitfall and funnel traps. Reptiles were also trapped in three unlogged stands that had not been burned or otherwise managed for 55 years.

(1) Greenberg C.H., Neary D.G. & Harris L.D. (1994) Effect of high-intensity wildfire and silvicultural treatments on reptile communities in sand-pine scrub. *Conservation Biology*, 8, 1047–1057.

7. Threat: Human intrusions and disturbance

Background

In addition to large-scale disturbances from activities such as agriculture, building developments, energy production and biological resource use, disturbance of reptile populations can come from smaller scale human intrusions such as recreation, pet entry into habitat areas or through human rock displacement in search of reptiles (Pike *et al.* 2010).

For studies that explore the potential for eco-tourism to reduce the threat posed to reptiles by human disturbance, see *Education and awareness raising - Offer reptile-related eco-tourism to improve behaviour towards reptiles*. Interventions to address predation by pets are included in *Threat: Invasive and other problematic species*.

See also *Threat: Biological resource use - Regulate wildlife harvesting*, and *Habitat restoration and creation - Maintain, create or restore rock outcrops*.

Pike D.A., Croak B.M., Webb J.K. & Shine R. (2010) Subtle–but easily reversible–anthropogenic disturbance seriously degrades habitat quality for rock-dwelling reptiles. *Animal Conservation*, 13, 411–418.

7.1. Use signs and access restrictions to reduce disturbance

- **One study** evaluated the effects on reptile populations of using signs and access restrictions to reduce disturbance. This study was in Turkey[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Reproductive success (1 study):** One replicated, controlled study in Turkey[1] found that in an area with signs where sea turtle nests were fenced, nests had higher hatching success than nests from areas with no fencing or signs.

BEHAVIOUR (0 STUDIES)

Background

The effects of summer (e.g. Moore *et al.* 2006) or winter (Sato *et al.* 2013) human recreational activity on reptiles can be negative (Larson *et al.* 2016). Different reptile species may tolerate different levels of disturbance, but it may lead to behaviour changes (Nyhof *et al.* 2015), nest abandonment (Moore *et al.* 2006), physical damage to reptiles or nests, or the displacement of individuals or populations. For vulnerable species or nests it may be possible to reduce the impacts of human disturbance using signs or access restrictions in areas subject to high use. Reducing access may also help reduce the risk of human introduction of non-native plants, animals or disease.

Larson C.L., Reed S.E., Merenlender A.M. & Crooks K.R. (2016) Effects of recreation on animals revealed as widespread through a global systematic review. *PLoS One,* 11, e0167259.

Sato C.F., Wood J.T. & Lindenmayer D.B. (2013) The effects of winter recreation on alpine and subalpine fauna: a systematic review and meta-analysis. *PloS one*, 8, e64282

Nyhof P.E. & Trulio L. (2015) Basking western pond turtle response to recreational trail use in urban California. *Chelonian Conservation and Biology*, 14, 182–184.

Moore M.J. & Seigel R.A. (2006) No place to nest or bask: effects of human disturbance on the nesting and basking habits of yellow-blotched map turtles (*Graptemys flavimaculata*). *Biological Conservation*, 130, 386–393.

A replicated, controlled study in 2000 on a sandy beach in southwest Turkey (1) found that sea turtle nests protected from human foot traffic using fencing and signs around individual nests tended to have higher hatching success rates than unprotected nests. Results were not statistically tested. Nests fenced for protection had 76% hatching success (667 of 880 eggs hatched, of which 653 hatchlings reached the sea) compared to 65% hatching success of unfenced nests (3,317 of 5,075 eggs hatched, of which 3,078 hatchlings reached the sea). All nests (12 nests) in a 2.5 km section of the 8 km long Fethiye Beach were fenced (70 × 70 × 150 cm with a 1 cm plastic mesh) with a sign "Do not disturb the turtle nests" in both Turkish and English to prevent human disturbance. Nests on the rest of the beach (72 nests) were unfenced. Nests were monitored from June to September 2000.

(1) Başkale E. & Kaska Y. (2005) Sea turtle nest conservation techniques on southwestern beaches in Turkey. *Israel Journal of Ecology and Evolution*, 51, 13–26.

7.2. Introduce and enforce regulations for reptile watching tours

- We found no studies that evaluated the effects on reptile populations of introducing and enforcing regulations for reptile watching tours.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Introducing regulations for reptile watching tours may reduce disturbance. This may involve setting limits on group sizes, observation times and the number of visits each day, particularly when tours involve visiting remote islands by boat. Interactions with reptiles during tours, such as feeding, may also be regulated. Enforcement of regulations may be required if compliance is low (e.g. Whitt & Read 2006).

Whitt A.D. & Read A.J. (2006) Assessing compliance to guidelines by dolphin-watching operators in Clearwater, Florida, USA. *Tourism in Marine Environments*, 3, 117–130.

7.3. Use nest covers to protect against human disturbance

- **Two studies** evaluated the effects of using nest covers to protect against human disturbance on reptiles. One study was in the USA[1] and one was in Greece[2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Reproductive success (2 studies):** One of two replicated, controlled studies (including one paired study) in the USA[1] and Greece[2] found that loggerhead turtle nests that were covered with cages had similar hatching success compared to nests that were not covered. The other study[2] found mixed effects of cages on hatching success of loggerhead turtle nests.

BEHAVIOUR (0 STUDIES)

Background

Human recreational activity may cause damage to reptile nests. Cage covers over individual nests may be used to prevent damage in areas of high human traffic.

For studies that discuss the use of nest covers to protect against predation, see *Threat: Invasive and other problematic species – Remove or control predators using fencing and/or aerial nets*.

A replicated, controlled, paired study in 1996 in beaches in Florida, USA (1) found that covering loggerhead *Caretta caretta* turtle nests with individual cages did not improve hatching success in areas with high or low human footfall. Hatching success was similar between caged and uncaged nests in areas of high footfall (caged: 66–67%, uncaged: 66–71%) and low footfall (caged: 75–76%, uncaged: 66–76%). In May–October 1996, fifty-eight paired sea turtle nests were either uncovered or covered with square wire cages (76 cm square, 107 cm tall, 5 x 10 cm mesh) anchored 30 cm in the sand in both low (66 total nests, 4,209 caged eggs, 3,888 uncaged eggs, 20 beach users/hour, two beach zones) and high traffic beaches (50 total nests, 3,678 caged eggs, 4,991 uncaged eggs, 50 beach users/hour, two beach zones. Hatching success was determined by excavating nests three days after hatchlings emerged to count successfully hatched eggs.

A replicated, controlled study in 1987–1995 on a sandy beach on Zakynthos Island, Greece (2) found that covering loggerhead turtles *Caretta caretta* nests with individual metal cages resulted in variable hatching success compared to both uncaged nests left in situ and nests relocated to an on-beach hatchery. Over six years, hatching success in caged nests was lower in two years, higher in two years and similar in two years compared to in situ nests. Hatching success for caged nests varied from 44–72%, compared to 56–68% for uncaged in situ nests and 51–75% for nests moved to an on-beach hatchery. From 1988–1995, nests located within 7 m of the sea and in danger of inundation were moved to a beach hatchery (77 nests) as were nests located near invasive plants which had root systems that grow into nests. From 1990, nests located in beach areas with tourists were protected by 50 cm circular metal mesh cages buried 15 cm in the sand (88 nests). A further 313 nests were left uncaged and in situ. Nests were excavated following hatchling emergence to assess hatching success.

(1) Mroziak M.L., Salmon M. & Rusenko K. (2000) Do wire cages protect sea turtles from foot traffic and mammalian predators? *Chelonian Conservation and Biology*, 3, 693–698.

(2) Kornaraki E., Matossian D.A., Mazaris A.D., Matsinos Y.G. & Margaritoulis D. (2006) Effectiveness of different conservation measures for loggerhead sea turtle (*Caretta caretta*) nests at Zakynthos Island, Greece. *Biological Conservation*, 130, 324–330.

8. Threat: Natural system modifications

Background

This chapter includes interventions to address threats that convert or degrade habitat as part of the management of natural or semi-natural systems, often to improve human welfare. This includes suppressing or increasing the intensity of fires, management of ground or mid-storey vegetation, changing the natural flow of water and managing beach erosion.

Fire and fire suppression

8.1. Use prescribed burning

Background

Fire is an integral part of the management and natural dynamics of some ecosystems. Some habitats are naturally fire-prone, and others have been shaped by long-term use of prescribed burning (Bowman 1998). Prescribed burns are undertaken to reduce the amount of combustible fuel in an attempt to reduce the risk of more extensive, potentially more damaging, 'wildfires'. They may also be used in the maintenance or restoration of habitats historically subject to occasional wildfires that have been suppressed through management or with the expressed purpose of enhancing wildlife habitat (Russell *et al.* 1999). Whilst burning can have a dramatic effect on the landscape, reducing cover and short-term food resources, the intensity of the fire may influence the response of reptiles to the prescribed burn–low-intensity fires may reduce shelters and prey availability while high-intensity fires may increase prey food items but reduce over wintering sites (Pearson *et al.* 2005). The impact of prescribed burning on habitats and their associated reptile populations are likely to vary depending on whether the vegetation is dominated by woody species, or by grasses and other herbaceous plants. As such, the impact of prescribed burning on reptile populations may vary in different habitats.

Due to the number of studies found, this action has been split by habitat type.

For studies that assess the affect of burning in combination with other actions see *Use prescribed burning in combination with vegetation cutting*; *Use prescribed burning in combination with herbicide application* and *Use prescribed burning in combination with grazing*.

Bowman D.M.J.S. (1998) Tansley Review No. 101. The impact of Aboriginal landscape burning on the Australian biota. *New Phytologist*, 140, 385–410.

Pearson D., Shine R. & Williams A. (2005) Spatial ecology of a threatened python (*Morelia spilota imbricata*) and the effects of anthropogenic habitat change. *Austral Ecology*, 30, 261–274.

Russell K.R., van Lear D.H., & Guynn Jr D.C. (1999) Prescribed fire effects on herpetofauna: Review and management implications. *Wildlife Society Bulletin*, 27, 374–384.

Forest, open woodland & savanna

- **Twenty-eight studies** evaluated the effects of using prescribed burning in forest, open woodland and savanna on reptile populations. Twenty-four studies were in the USA[1,3-7,9-14,16-21,23-28], three were in Australia[2,8,15] and one was in Brazil[22].

COMMUNITY RESPONSE (12 STUDIES)

- **Community composition (1 study):** One replicated, randomized, controlled, before-and-after study in the USA[16] found that in areas with prescribed burning, reptile assemblages became similar to more pristine areas that had historically experienced frequent fires.
- **Richness/diversity (11 studies):** Seven studies (including two replicated, randomized, controlled, before-and-after studies) in the USA[6,9,11,13,23,26] and Australia[2] found that burned areas had similar reptile species richness compared to unburned areas. One of the studies[6] also found that burned areas had higher reptile diversity than unburned areas. Two replicated studies (including one randomized, controlled study) in Australia[15] and the USA[25] found that reptile species richness remained similar with time since burning. One of two studies (including one replicated, randomized, controlled, before-and-after study) in the USA[1,20] found that burned areas had higher combined reptile and amphibian species richness[1] than unburned areas. The other study[20] found that burned areas had similar combined reptile and amphibian species richness and diversity compared to unburned areas.

POPULATION RESPONSE (26 STUDIES)

- **Abundance (23 studies):** Nine of 21 studies (including four replicated, randomized, controlled, before-and-after studies) in the USA[1,3,5-7,9-11,13,14,17-21,23,24,26,27] and Australia[2,8,15] found that burning had mixed effects on the abundance of reptiles[2,3,5,8,14,18,27], six-lined racerunners[17] and western yellow-bellied racer snakes[19]. Six studies found that burned areas had a higher abundance of reptiles[1,6,9], lizards[7], black racer snakes[24] and more active gopher tortoise burrows[10] compared to unburned areas. The other six studies found that burned areas had a similar abundance of reptiles[11,13,23,26], lizards[20] and gopher tortoise burrows[21] compared to unburned areas. One replicated, site comparison study in Australia[15] found that reptile abundance increased with time since burning. One replicated, randomized, controlled study in the USA[25] found that burning in different seasons had mixed effects on the abundance of reptiles.
- **Survival (2 study):** One of two studies (one site comparison and one controlled study) in the USA[12] and Brazil[22] found that Texas horned lizard survival was similar in burned and unburned areas. The other study[22] found that burning had mixed effects on survival of an endemic lizard species.
- **Condition (1 study):** One site comparison study in the USA[28] found that eastern fence lizards in recently burned areas ran faster than those from areas that were burned less recently or were unburned.

BEHAVIOUR (2 STUDIES)

- **Behaviour change (2 studies):** One replicated, controlled, before and-after study in the USA[10] found that burning affected overwintering habitat use by gopher tortoises. One replicated, controlled study in the USA[24] found that in burned areas, black racer snakes had higher surface activity than in unburned areas.

A site comparison study in 1982–1984 of sandhill-scrub habitat in west central Florida, USA (1) found that controlled burns resulted in higher species diversity and abundance of combined reptiles and amphibians. Results were not statistically tested. The seven-year burn cycle plot had the greatest number of reptile and amphibian species in both years (7-year cycle: 16–20 species; 2-year: 10–15; 1-year: 14–16; unburned: 10–15). Although burn plots had greater fluctuations in species diversity over the two years than the unburned plot, numbers of captures were higher. Captures tended to be highest in seven- and one-year burn plots (7 years: 115–307 individuals; 2 years: 102–187; 1 year: 126–203; unburned: 71–125). The one-year cycle was most consistent for supporting high numbers of individuals and species. The six-lined race runner *Cnemidophorus sexlineatus* was the most abundant lizard and densities were greatest on one-year burn cycles (see paper for details). One ha plots were established for one-, two-, and seven-year burn cycles in adjacent strips with burns taking place from 1976. These were compared to a plot unburned for 20 years (last burn in 1965). Burns were in May–June. Five drift-fence arrays with pitfall traps and an artificial cover board were established/plot. Traps were checked 5–6 times/week in April–October 1983–1984.

A replicated, site comparison study in 1994 of native forest and non-native pine plantations near Brisbane, Australia (2) found that reptile abundance was higher in burned native forest than unburned forest, but lower in burned plantations than unburned, and that species richness was unaffected. Reptile-only results were not statistically tested. In native forest there were more reptiles captured in five-year burn cycles than unburned sites (5-year cycle: 60, 3-year: 40; unburned: 31). In pine plantations, fewer reptiles were found in burned sites than unburned sites (burned seven years ago: 16, burned two years ago: 5, unburned: 33). Species richness was similar between burned (3–8 species) and unburned plots (6–7 species). Treatments in native forest (1.5 ha; two replicates) were: burned in autumn–winter on a three-year cycle (burned 1991), in winter–spring on a five-year cycle (burned 1993) or unburned (since 1973). In the plantation (25 ha) treatments were: burned two or seven years ago, or unburned. Drift-fencing with pitfall traps and active searching were used for monitoring in January or March 1994 (75–180 trap nights/treatment).

A replicated, site comparison study in 1992–1993 in mixed hardwood and pine coastal forest in Maryland, USA (3) found that annual prescribed burning did not increase reptile abundance. Following 4–5 years of annual prescribed burning, overall reptile and snake, but not skink or turtle, abundances were reduced in burned pine plots (overall: 96 individuals, snake: 65, skink: 31, turtle: 0) compared to unburned mixed pine-hardwood plots (overall: 130, snake: 91, skink: 36, turtle: 3). See paper for individual species comparisons. The numbers of reptiles captured in burned plots (96 individuals) tended to be lower than in unburned hardwood forest (200 individuals, results were not statistically tested). In March–July 1992–1993, reptiles were monitored in three locations each in four forest stands: prescribed burn pine (4 ha total area), unburned mixed pine-hardwood (5 ha), unburned hardwood (328 ha) and unburned, cut hardwood (130 ha). One third of the burned pine area was burned annually (a different section each year) since 1988. Prior to this it was burned annually in 1981–1984.

Surveys were carried out using drift fences with pitfall and funnel traps ('arrays', 1992: 366–381 array nights/stand; 1993: 423).

A replicated, controlled, before-and-after study in 1995–1996 in mixed hardwood and pine forest in North Carolina, USA (4) found that prescribed burning did not tend to affect the abundance of reptiles (although very few reptiles were captured overall). Prior to burning, no reptiles were captured in the burn sites and one northern ringneck snake *Diadophis punctatus* was captured in a site that was not to be burned. After burning, one five-lined skink *Eumeces fasciatus* and one eastern garter snake *Thamnophis sirtalis* were captured in burned plots in the same year as the burn took place. Monitoring was undertaken for two weeks immediately before an April burn and after the burn in June 1995 and August 1996 at two sites in a 1,820 ha area of national forest. Drift-fencing with pitfalls and snap-traps were installed at three locations in the upper slope, mid-slope and riparian zone at each site. Visual searches were also undertaken. An unburned area at one of the sites was monitored in the same way.

A replicated, randomized, controlled study in 1997–1998 of pine sandhills in Florida, USA (5) found that prescribed burning had mixed effects depending on species and year. In one of two burn years, capture rates of six-lined racerunners *Cnemidophorus sexlineatus* and eastern fence lizards *Sceloporus undulatus* were higher in burned compared to fire-suppressed plots (burned: 0.007–0.037 captures/trap days; fire suppressed: 0.002–0.015) and southeastern crowned snake *Tantilla coronata* capture rates were lower in burned than in fire suppressed plots (burned: 0.004; fire suppressed: 0.014 captures/trap days). Green anole *Anolis carolinensis* were captured at similar numbers in burned and fire suppressed plots (0.003 captures/trap days in 1998 for both treatments). Plots (81 ha) were randomly selected for burning (4 plots) or continued fire-suppression (4 plots) and burning took place in spring 1995. Monitoring was undertaken using drift-fencing and pitfall traps in April–August 1997–1998.

A replicated, site comparison study in 2001 in mixed hardwood and pine forest in Georgia, USA (6) found that reptile abundance and diversity, but not species richness, were higher in burned compared to unburned sites. Reptile abundance and diversity but not species richness were greater in burned stands (abundance: 6.7 individuals/stand, diversity: 0.4 Shannon-Wiener Index, richness: 3 species/stand) compared to unburned stands (abundance: 4.3, diversity: 0.2, richness: 2.3). In total, 21 individuals of 8 species (5 lizards, 3 snakes) were captured in burned stands compared to 13 individuals of 4 species (3 lizards, 1 snake) in unburned forest stands. In July–October 2001, reptiles were monitored in three burned (every 2–3 years for 9 years, most recently in January 2001) and three unburned forest stands. Surveys were carried out using drift fences with pitfall traps, coverboards and PVC pipes (348 survey nights/stand). Burns were carried out during winter and did not significantly affect coarse woody debris volumes, but did reduce leaf litter depth.

A replicated, randomized, controlled study in 1995–1996 in oak forests in Virginia, USA (7) found that more lizards were captured, but there were similar numbers of reptile species in burned compared to unburned sites. One year after burning, lizards were captured more often in burned (2–3 individuals/plot) than unburned forest (1 individuals/plot). One snake was caught each in burned and unburned plots (see original paper for individual species capture rates). Overall reptile richness was 2–4 species in burned and 3 species in unburned plots (results were not statistically tested). Plots in three forest stands (2–5 ha) were burned in February, April or August 1995 in a randomized block design or left unburned. All plots had been subject to shelterwood harvest 3–5 years before burning. Reptiles were monitored using pitfall traps one year after burning for 53 nights during June–October 1996 (12,720 total trap nights).

A replicated, site comparison study in 2001 in three sites within savanna woodlands in Queensland, Australia (8) found that overall reptile abundance was similar in burned and unburned areas, though the abundance of one species was higher after burning and another species abundance was lower after burning in combination with grazing. Overall reptile abundance was similar in burned (12–20 individuals/plot) and unburned plots (14–19), regardless of grazing practices. Of 18 species recorded, one dragon species abundance was higher in burned than unburned plots regardless of grazing and one ctenotus abundance was lower in burned than unburned plots, particularly when burning was combined with grazing (central netted dragon *Ctenophorus nuchalis* burned: 0.7–1.0 individuals/plot vs. unburned: 0–0.1; leopard ctenotus *Ctenotus pantherinus* 0–1 vs. 1–4). In January 2001, reptiles were monitored on three cattle stations (>20,000 ha each) in 29 one-ha plots that were either recently burned (within 2 years) or unburned (last burnt >2 years ago) and either ungrazed (paddocks where cattle were excluded) or grazed (4–8 cattle/ha). Burns were a mixture of prescribed burns and wildfires and all treatments took place over >2,000 ha areas. Reptiles were sampled using cage traps and pitfalls supplemented by day and night log rolling and litter raking.

A replicated, controlled study in 2003–2004 of pine savanna in Mississippi, USA (9) found that after prescribed burning reptile abundances tended to be higher, but species richness was similar compared to unburned sites. Results were not statistically tested. In burned sites, 1.3 individuals/transect and 5.0 species/site were captured, compared to 0.9 individuals/transect and 5.3 species/site in unburned sites. A low intensity burn was undertaken over a large proportion of a National Wildlife Refuge in March 2003. From January to June 2004, reptiles were monitored at three burned and three unburned sites. Visual encounter surveys (200 m transects), minnow traps (six/site) and PVC tubes (five/site) were used.

A replicated, controlled, before and-after study in 2002–2004 in mixed open shrub and forest habitat in Mississippi, USA (10) found that proportions of active gopher tortoise *Gopherus polyphemus* burrows tended to be higher in forest areas after burning, and that after burning of both forests and forest clearings, more tortoises overwintered in the clearings compared to when forests and clearings were left unburned. Results were not statistically tested. Two years after prescribed burning began, 33 of 34 (97%) gopher tortoise burrows in burned forest were active, compared to 11 of 16 (69%) active burrows before burning. In unburned sites proportions of active burrows in forests were 82–88% over the same period (2002: 14 of 17 active burrows; 2004: 23 of 26 active burrows). In burned sites, 84% of tortoises overwintered in open shrubland and 16% in forest-interior burrows, compared to 59% overwintering in open shrubland, 27% in forest-interior and 14% in forest-edge burrows in unburned sites. Tortoises spent similar amounts of time in open shrubland versus forest habitat and similar hibernation durations in burned and unburned sites (see original paper for details). Four of eight forest and shrubland sites were burned in January–February 2002 and April 2003. Tortoise burrows were surveyed in April 2002–2004. Tortoises (4–7 individuals/site, 20 in burned sites, 20 in unburned sites) were trapped and monitored using radio-telemetry twice a week in the active seasons and once a week in the dormant seasons (>70 times/year/individual).

A replicated, randomized, controlled, before-and-after study in 2001–2004 in an upland hardwood forest in North Carolina, USA (11, same experimental set-up as 26) found that burned and unburned areas had similar abundance and species richness of reptiles. Reptile species richness was similar in burned and unburned areas (1–2 species). Total reptile abundance was also similar in burned (1–3 reptiles/100 nights) and unburned (3 reptiles/100 nights). See original paper for other individual species abundances. Three forest segments were divided into different management zones (14 ha each): prescribed burn and unburned. Reptiles were surveyed using drift fence arrays with pitfall and funnel traps before any burning took place in August–October 2001 and after burning in May–September 2002–2004.

A site comparison study in 1998–2001 in thornscrub in southern Texas, USA (12) found Texas horned lizard *Phrynosoma cornutum* survival was similar in prescribed burned and unburned sites. Four-month survival rates of Texas horned lizards were similar between burned and unburned sites that were also subject to livestock grazing (burned: 47%, unburned: 35%). Lizard survival rates were initially higher in the May–June in the second year after burning than in the first year after burning, but overall annual survival rates were similar between the two years (second year after burning: 49%, first year: 32%). Lizard survival was monitored in burned and unburned sites (50–60 ha each) in a wildlife management area (6,500 ha). Lizards were captured by searching roads, chance encounters and drift fences with pitfall traps. Lizards were marked with a PIT tag and toe clips, and fitted with a radio transmitter. Lizards were located at least once every 24 hours for four months from mid-April to mid-August in 1998–2001 (burned sites: 48 lizards, unburned sites: 39 lizards).

A replicated, randomized, controlled study in 2006–2007 in hardwood forests in North Carolina, USA (13) found that burned areas had similar overall reptile species richness and capture rates compared to unburned areas. Overall reptile richness and overall reptile, snake and turtle captures were similar in burned areas (richness: 4–7 species/100 array nights, overall captures: 7–9 individuals/100 array nights, snakes: 2–7 individuals/100 array nights, turtle: 0–1 individuals/100 array nights), and unburned areas (6, 7–7, 3–5, 0). Three sites (10 ha) each were managed by twice-burning (in March 2003 and February 2006) or received no management ('unburned'). Reptiles were surveyed in May – August 2006 and 2007 using drift fences with pitfall traps ('arrays', 3/site).

A replicated, randomized study in 1999–2001 in nine restored pine woodlands in western Arkansas, USA (14) found that overall reptile captures did not change during a three-year burning cycle, but some individual species capture rates varied with time after burning. Captures were similar in all years of a three-year burn cycle for overall reptiles (year 1: 79 individuals/stand, year 2: 79, year

3: 76) snakes (31, 33, 39), lizards (47, 44, 36) and turtles (0.7, 1.4, 0.9). Southern black racer snake *Coluber constrictor priapus* captures were lowest in the burn year (3 individuals/stand) compared to the two subsequent years (7–9). Ground skink *Scincella lateralis* captures were highest in the burn year (16 individuals/stand) compared to the two subsequent years (7–9). Southern coal skink *Eumeces anthracinus pluvialis* captures were highest in the second year after burning (0.7 individuals/stand) compared to the previous two years (0.1–0.1) and fence lizard *Sceloporus undulatus* captures were higher in the year after burning (19 individuals/stand) compared to the burn year (11) but similar to the second year after burning (13). In 1999–2001, nine stands (11–42 ha) were burned on a three-year cycle, so three were burned each year in March–April. Stands had been thinned at least nine years previously and had undergone 3–7 prescribed burns at 2–5-year intervals. Monitoring was undertaken using three drift-fence arrays/stand (15 m) connected to central funnel traps in April–September in 1999–2001.

A replicated, site comparison study in 2008–2011 of 74 temperate woodland sites Victoria, Australia (15) found that reptile abundance but not species richness increased with time since fire. Reptile abundance was lower in sites that were burned more recently and higher in sites with a longer time since burning (results reported as model outputs, see paper for details). Species richness was similar in recently burned sites compared to long-term unburned sites. A total of 2,691 reptiles of 14 species (10 lizards and 4 snakes) were captured. In summer 2008–2011, reptiles were surveyed in 74 sites ranging recently burned (0 years since fire) to approximately 80 years post-burn. Fire histories included both prescribed burning and wildfires. Surveys were carried out using drift fences with pitfall traps (14,084 total trap nights). Sites were used for a maximum of two seasons.

A replicated, randomized, controlled, before-and-after study in 1995–2010 in fire-suppressed longleaf pine *Pinus palustris* forest in Florida, USA (16, same experimental set-up as 17) found that after regular prescribed burning to remove hardwood trees, reptile assemblages became similar to more pristine sites that had historically experienced frequent fires. All results reported as statistical model outputs, see original paper for details. After 10 years of regular prescribed burning to remove invasive hardwood trees, reptile assemblages in prescribed burning sites were similar to sites that had historically experienced frequent fires. See original paper for details of individual species responses to management. Reptiles were monitored in four sites (81 ha) each that were managed by prescribed burning (April–June 1995, four sites) or were unburned until after 1999 when all sites were burned at 2–3-year intervals. Reptiles were also monitored a further four sites in an area without historical fire suppression. Reptiles were surveyed using drift fences with pitfall traps (16 traps/site) in April–August 1997–1998 and May–September 2009–2010.

A replicated, randomized, controlled, before-and-after study in 1995–2010 in fire-suppressed longleaf pine *Pinus palustris* forests in Florida, USA (17, same experimental set-up as 16) found that areas with prescribed burning had similar six-lined racerunner *Aspidoscelis sextineatus* abundance compared to pristine areas, whereas unburned areas had fewer. In the first two years, six-lined racerunner abundances were similar in burned areas (adults: 23 individuals/site; juveniles: 10) and more pristine areas with a history of frequent fires (adults: 38, juveniles: 10), and higher than in unburned areas (adults: 13, juveniles: 2). After 15 years, when all sites were regularly burned, six-lined racerunner abundances were similar in sites that had been previously burned (adult: 40 individuals/site; juvenile: 7), unburned (30, 6) or in more pristine areas with a history of frequent fires (37, 10). Reptiles were monitored in six plots (81 ha) each that were burned (April–June 1995, 6 plots) or unburned (6 plots) until after 1999 when all plots were burned at 2–3-year intervals. Reptiles were also monitored in a further six plots in more pristine areas with a history of frequent fires that was considered to be the target condition of restoration efforts. Reptiles were surveyed using drift fences with pitfall traps (16 traps/site) in April–August 1997–1998 and May–September 2009–2010.

A replicated, controlled, before-and-after study in 2005–2008 in mixed forest in Alabama, USA (18) found that following burning, the abundance of one reptile species increased and seven remained similar. Eastern fence lizard *Sceloporus undulatus* captures increased after burning (pre-burn: 0 individuals/100 trap nights, post-burn 1–4). The abundance of seven other species was not affected by burning (see paper for details). In 2005–2008, the impact of burning compared to no management on reptiles was tested (three 9 ha plots/treatment). Reptiles were surveyed for 3–6 months before burning began (April–August) and in the two years after burning (in March–September) using drift fences with pitfall traps.

A controlled study in 2005–2010 in a mixed coastal wetland, scrub and woodland habitat in California, USA (19) found that four years after prescribed burning, western yellow-bellied racer snake *Coluber constrictor mormon* abundance was lower in burned than unburned sites, but that abundance was similar in burned and unburned sites from five years after burning took place. Four years after prescribed burning, western yellow-bellied racer snake abundance was lower (2008: 17 snakes/trap array) compared to unburned sites (49). In the fifth and sixth years after burning, snake abundance was similar in burned and unburned sites (2009 burned: 16 snakes/trap array vs. unburned: 25; 2010 burned: 19 vs. unburned: 30). Prescribed burns were carried out in a 213 ha area in autumn 2005 (64 ha) and 2006 (67 ha). Reptiles were surveyed in burned and adjacent unburned areas using traps, observation and coverboards. Traps were set in March–August 2007–2010 (277–1,140 trap days/year). Caught snakes (692 total individuals) were individually marked using PIT tags. Too few individuals were caught in the 2006 burn site to be included in analysis.

A replicated, randomized, controlled, before-and-after study in 1999–2007 in six pine plantations in Mississippi, USA (20) found that prescribed burning did not increase reptile and amphibian richness, diversity or species abundances, apart from one lizard species in one of seven years. In six of seven years after burning, species richness, diversity measures and species abundances were similar in burned and unburned plots (data reported as model outputs, see paper for details). Eastern fence lizard *Sceloporus undulatus* abundance was higher in burned plots (0.02 lizards/plot) in the first year after management compared to unburned plots (0.002 lizards/plot). Six plots each (10 ha plots) in six intensively-managed, 18–22-year-old commercial pine stands (59–120 ha) were burned or left unburned. Burning took place in the dormant season (December–February) in 2000, 2003 and 2006. Reptiles were monitored using drift fences with pitfall and funnel traps in May–June 1999–2007 (one year before management and seven years after management began).

A replicated, site comparison study in 2005–2011 in two sites of beach dunes, dry hammock and freshwater marsh in Florida, USA (21) found that gopher tortoise *Gopherus polyphemus* burrow density tended to be similar in areas that were burned or unburned. Results were not statistically tested. Burrow density ranged from 0.6–0.8 burrows/ha in burned areas and 0.6–0.7 burrows/ha in unburned areas. The authors suggest there may have been increases in burrow density from 2005–2011 in areas burned at least once since 2005 (0.2 to 1.2 burrows/ha; 0.3 to 0.5 burrows/ha; 7.5 to 10.3 burrows/ha) and a decrease in an area not burned since 2005 (2.7 to 0.2 burrows/ha). Four areas of the site had a history of prescribed burns dating back to 1988 (160 ha total), whereas other areas had no history of burning (229 ha total). Three of the burned areas were burned at least four times since 1988, and one was burned only once since 2010, and three of four areas were burned at least once during the study period. Burrow surveys were conducted in 2005, 2007 and 2011 by groups of surveyors walking 5–10 m apart during spring and autumn.

A controlled study in 2005–2013 in neotropical savanna in Brasilia, Brazil (22) found that regular prescribed burning increased adult and juvenile endemic lizard *Micrablepharus atticolus* survival in the short term, but that more frequent late-dry season burns were detrimental. All results were reported as statistical model outputs, see original paper for details. In months when prescribed burns took place, lizard survival and recruitment rates increased. Lizard survival rates were lowest in the late-season biennially burned plot, but similar in plots burned in early-dry season, mid-dry season, or not prescribed burned. Juvenile survival was lowest in biennially-burned plots and highest in the unburned plot. In November 2005 to March 2013, five plots (10 ha each) in an ecological reserve were sampled for lizards. Plots were prescribed burned in either: early-dry season (June) biennially, mid-dry season (August) biennially, late-dry season (September) biennially, mid-dry season quadrennially, or not prescribed burned (although burned in an unplanned fire in September 2011). Lizards were sampled daily using a drift fence with pitfall traps in each plot (see original paper for details) for six consecutive days/month. Lizards were individually marked by toe clipping, measured, sexed and released (465 individual lizards were caught during the study).

A replicated, randomized, controlled, before-and-after study in 2008–2014 in an upland mixed oak forest in the Appalachians, USA (23) found that prescribed burning did not increase the abundance or species richness of total reptiles, snakes or lizards when compared to unburned areas. Total reptile and snake species richness was similar in prescribed burn areas and unburned areas (data presented as statistical model results), as was lizard species richness (burned:0–0.4 species/plot; unburned: 0–0.8 species/plot). Abundance of total reptiles was similar in prescribed burn areas (0.1–0.3 average captures/100 fence nights) compared to unburned areas (0.2–0.5 average captures/100 fence nights). Prescribed burn plots and unburned plots (five 5 ha plots of each in 2008, 2010–2011 and four plots of each in 2013–2014) were monitored using drift fences, pitfall and funnel traps in May to August one year pre-treatment (2008) and five years post treatment (sampled in 2010, 2011, 2013, 2014; see paper for details of trap deployment).

A replicated, controlled study in 2012 in mature oak-hickory forest in south-western Kentucky, USA (24) found that in burned plots, black racer snake *Coluber constrictor* abundance was higher and snakes increased surface activity compared to unburned plots. Snake abundance was higher in burned plots (39 individuals) compared to unburned plots (21 individuals). Snakes were more active on the surface in burned plots than unburned plots (data presented as statistical model outputs). Males moved more than females in burned plots and less than females in unburned plots although sex ratios and body sizes were similar between burned and unburned plots (see original paper for details). Radio-tracked snake mortality rates were higher in burned areas (5 individuals) than unburned areas (1 individual, no statistical tests were carried out). Data was collected from two treatments: burned (in April 2007 and September 2010) and unburned plots. Snakes were trapped and monitored in eight square 64 ha plots (four burned, four unburned). Plots were > 200 m from the treatment edge and at least 500 m from the nearest plot. Snakes were trapped using drift fences, funnel and pitfall traps during April–August in 2012. Snake movements were monitored using radio transmitters (burned: 11; unburned: 10).

A replicated, randomized, controlled study in 2013–2016 in one oak-dominated forest in North Carolina, USA (25) found that reptile species richness did not change after prescribed burning, but that capture rates were higher and decreased over time after growing-season, but not after dormant-season prescribed burning. Reptile species richness did not change over time following dormant-season or growing-season burns, or in unburned plots (data reported as model results). Reptile capture rates were highest in the year of growing-season burns (8 individuals/100 array nights; 1–3 years later: 3–6 individuals/100 array nights), but did not change after dormant-season burns (1 individuals/100 array nights; 1–2 years later: 1–2 individuals/100 array nights) or in unburned areas (1–2 individuals/100 array nights). There was no difference in overall lizard or overall snake capture rates between burned (growing season and dormant season) and unburned plots over time. See original paper for species-specific capture rates. The authors reported that growing-season burns cleared more canopy cover than dormant-season burns, which may have contributed to elevated reptile captures in growing-season burned plots compared to unburned plots. Nine plots (4–7 ha) in a National Forest were either prescribed burned in the growing season (April 2013), dormant season (March 2014), or not burned (three plots/approach). Reptiles were surveyed using drift fences with pitfall traps ('arrays') May–August 2013–2016 (2–3 arrays/plot, dormant-season burn plots only surveyed in 2014–2016).

A replicated, randomized, controlled study in 2001–2016 in upland forest in North Carolina, USA (26, same experimental set-up as 11) found that burned areas had similar overall species richness and individual species abundance compared to unburned areas. Overall reptile species richness was similar between prescribed burning and unburned forest (data reported as model outputs). In 2016, five-lined skink *Plestiodon fasciatus* and eastern fence lizard *Sceloporus undulatus* capture rates were similar in burned and unburned areas (skink - burning: 2.8 skinks/100 trap group nights; unburned: 1.6; lizard - burning: 3.6;

unburned: 0.5). Three study sites were selected within a 5,840 ha mixed oak-hickory forest. Within each site, experimental plots (10 ha core areas with 20 m wide buffers) were burned (2003, 2006, 2012, 2015) or left unburned. Reptiles were surveyed after burns using drift fences with pitfall and funnel traps in May – August of 2003–2004, 2006–2007, 2014 and 2015–2016 (158–341 trap group nights/plot/year).

A replicated, controlled study in 2007–2016 of an oak/hickory forest in western Kentucky, USA (27) found that overall snakes, two snake species and one lizard species, but not overall lizards were more abundant in prescribed burned areas compared to unburned areas. Abundance of snakes overall was significantly higher in burned than unburned plots, but abundance of lizards overall was not (data reported as model outputs). North American racer snakes *Coluber constrictor*, ring-necked snakes *Diadophis punctatus* and eastern fence lizards *Scelophorus undulatus* were more abundant in burned (racer snakes: 4–13; ring-necked snakes: 4–15; fence lizards: 11–45 individuals/100 fence nights) than unburned plots (racer snakes: 1–9; ring-necked snakes: 1–3; fence lizards: 5–18 individuals/100 fence nights), although the size of the effect varied by year. Abundance of all reptiles and reptile community structures were similar in burned and unburned plots (data reported as model outputs, see original paper for details). Prescribed burns took place in 2007 and 2010. Data were collected in four burned and four unburned 800 x 800 m study areas in spring and summer 2011, 2012, 2015 and 2016. Reptiles were surveyed using drift fencing, pit fall and funnel traps. Reptiles captured included five snake and four lizard species.

A site comparison study in 2014–2015 in mixed oak-hickory forest in Kentucky, USA (28) found that in prescribed burn forest eastern fence lizards *Sceloporus undulatus* ran faster than those in forests that had not been exposed to fire for four years or were unburned. Eastern fence lizards from forest burned less than six months previously ran faster (maximum sprint speed: 3.1 m/second; 2 m run speed: 2.2 m/second) than eastern fence lizards from forest burned four years previously (maximum sprint speed: 2.6 m/second; 2 m run speed: 1.7 m/second) or unburned forest (maximum sprint speed: 2.3 m/second; 2 m run speed: 1.6 m/second). In 2014, eighty lizards were captured, measured and speed tested. Similar size and weight lizards were captured from forest that had been prescribe burned less than six months earlier (26 lizards), or four years earlier (26 lizards), or not burned in the previous 60 years (28 lizards). Lizards were placed on a track and encouraged to run at maximum speed. Lizard top sprint speed and running speed over 2 m were measured using video technology (see original paper for details). Fourteen lizards were recaptured and retested in 2015.

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Grassland & shrubland

- **Fourteen studies** evaluated the effects of using prescribed burning in grassland and shrubland on reptile populations. Seven studies were in the USA[2,4,5,7-10], four were in Australia[1,11,12,14] and one was in each of South Africa[3], Argentina[6] and France[13].

COMMUNITY RESPONSE (6 STUDIES)

- **Community composition (1 study):** One replicated, before-and-after study in Australia[11] found that reptile species composition was different before and immediately after burning in three grass types and remained different after vegetation grew back in one of three grass types.

- **Richness/diversity (5 studies):** Two of three studies (including one replicated, controlled, before-and-after study) in South Africa[3], Argentina[6] and the USA[9] found that areas with annual burning had similar reptile species richness and diversity compared to unburned areas[6] or that richness was similar across areas with a range of burn frequencies[3]. The other study[9] found that burned areas had higher reptile species richness than unburned areas. One replicated, site comparison study in Australia[1] found that areas burned 1–4 years earlier had lower reptile species richness than areas burned 11–15 years earlier. One replicated, site comparison study in the USA[2] found that areas with different burn frequencies had similar reptile species richness and diversity.

POPULATION RESPONSE (11 STUDIES)

- **Abundance (11 studies):** Three of six studies (including three replicated, randomized, controlled studies) in the USA[2,4,5,7,8] and Argentina[6] found that burned areas had a similar abundance of lizards[4], snakes and lizards[5] and combined reptiles and amphibians[2] compared to unburned areas. Two studies[6,7] found that burning had mixed effects on the abundance of different reptile species[6] and western yellow-bellied racer snakes[7].The other study[8] found that burned areas had more eastern massasauga rattlesnakes than unburned areas. One replicated, site comparison study in Australia[1] found that areas burned 1–4 years earlier had a lower abundance of reptiles than areas burned 11–15 years earlier. One controlled before-and-after study in the USA[10] found that a burned area had a similar number of four snake species compared to when the area was managed by mowing. One site comparison study in France[13] found that one reptile species was less abundant in areas managed by burning than areas grazed by sheep, whereas the abundance of five other species was similar in all areas. One replicated, before-and-after study in Australia[11] found that immediately after burning, the abundance of reptiles was lower than before burning, but was similar after vegetation grew back. One replicated, randomized, site comparison study in Australia[14] found that small-scale patch burning was associated with increased abundance of sand goanna burrows.

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One replicated, randomized, controlled study in Australia[12] found that some rocky outcrops that were burned were recolonized by pink-tailed worm-lizards.

A replicated, site comparison study in 1987–1990 in spinifex grasslands in the Northern Territory, Australia (1) found that nine months to five years after prescribed burning, overall reptile species richness and abundance were lower than plots that had been burned 11–15 years earlier. Regenerating plots burned 9 months –4 years earlier had lower overall reptile abundance and species richness (abundance: 1–24 individuals/plot, richness: 1–12 species/plot) compared to mature plots burned 11–15 years earlier (4–47, 2–15). The relative abundances of species on regenerating plots changed after the first year of sampling, with terrestrial geckos becoming less common and *Ctenotus* species more common, whereas relative abundances of reptiles changed little in mature plots (see original paper for details of species individual and relative abundances). In 1987–1990 reptiles were surveyed 12 times (approximately every three months) in plots that had been burned in 1986 ('regenerating') or in 1976 ('mature'; 3 plots/burn history). Surveys were carried out using drift fences with pitfall traps for three nights at a time (18 traps/plot).

A replicated, site comparison study 2003–2004 in six watersheds in tallgrass prairie in Kansas, USA (2) found that carrying out burning more frequently did not result in differences in combined reptile and amphibian abundance, species richness or diversity. Species richness and overall abundance were similar between areas with an annual burn (5–10 species/plot; 91 individuals), a four-year burn (6 species/plot; 115 individuals) or that remained unburned for 10–20 years (5–6 species/plot; 89 individuals), though the abundance of individual species in each treatment was mixed (see paper for details). Species evenness and diversity was similar across areas with different burn regimes, although reptile communities differed, with areas with more similar burning sharing more species (all results reported as indexes). Six watersheds were selected: two with annual burns; two burned every four years; and two that were unburned for 10–20 years. Two transects were established/watershed, and each transect (75 m long) incorporated cover boards, drift fencing (Y-trap array each end of transect) and funnel traps. In spring (1 month) and autumn (1 month) 2003–2004, traps were checked daily until temperatures reached 32°C and captured species were identified and marked.

A replicated, site comparison study in 2004 in an area of grassland in Gauteng, South Africa (3) found that reptile species richness was similar in areas that were last burned one, two or three years ago, and was also not affected by burn frequency over the previous 30 years. Neither time since last burn or frequency of burning over the past 30 years affected reptile species richness (data reported as statistical model result). In March–April 2004, reptiles were surveyed in nine sites that had last been burned one year ago (4 areas), two years ago (3 areas) or three years ago (2 areas). Burn frequency in the preceding 30 years of the sites varied from burning every 1–5 years. A total of 10 groups of traps (4 drift fences, 8 funnel and 8 pitfall traps) were established across the nine sites (1–2 groups/block). Traps were checked twice/day and all reptiles were identified to species level.

A replicated, randomized, controlled, paired study in 2004 in juniper and mesquite shrublands in central Texas, USA (4) found that after prescribed burning, lizard captures were similar to unburned plots. Lizard captures were statistically similar in burned (8 lizards/plot) compared to unburned plots (4 lizards/plot). Low intensity prescribed burns were carried out in 0.2 ha plots, paired with unburned plots, in four locations on a former livestock ranch in February 2004. Lizards were surveyed using arboreal pitfall (1 m long, 8 cm diameter PVC tubes), glueboards (stapled to trees) and drift fences (one/plot) and terrestrial glueboards (a 4 x 4 grid 5 m around each arboreal trap) for four trapping sessions in March–August 2004 (152 traps and 5,908 total trap nights).

A replicated, randomized, controlled study in 1997–2001 in shrub and grassland in southern Texas, USA (5) found neither dormant (winter) season nor growing (summer) season prescribed burns affected the abundance of lizards or snakes in subsequent years. In the 2–3 years after dormant-season and growing-season prescribed burns, abundances of lizards and snakes in burned plots (dormant-season: 0.4–1.1 lizards/trap array/day, 0.1–0.3 snakes/trap array/day; growing-season: 18.2–19.8 lizards/trap array/day, 2.4–4.0 snakes/trap array/day) were similar to unburned plots (dormant-season: 0.8–1.4 lizards/trap array/day, 0.1–0.2 snakes/trap array/day; growing season: 13.6–14.8 lizards/trap array/day, 1.0–2.2 snakes/trap array/day). Dormant-season (December–February 1997–1998, 1999–2000) and growing-season (August 1999) prescribed burns were carried out in 2 ha plots (dormant: 3 plots, growing: 5) in a 15,200 acre study area. Reptiles were monitored using drift fences with pitfall traps ('arrays', dormant: 3 arrays/plot, growing: 1 array/plot). Dormant-season plots were monitored for 7–21 days each in May–August 1998–2000. Growing-season plots were monitored for 14 days each in May–September 2000–2001. Equivalent numbers of unburned plots were monitored at the same time. Prior to 1997, dormant-season plots had not been burned for ≥40 years. Growing-season plots had been prescribed burned in January–March 1997.

A site comparison study in 2006 of cattle pasture in Corrientes, Argentina (6) found that overall reptile diversity, species richness and abundance were similar in areas with annual burning and unburned areas. Overall reptile species richness, abundance and diversity were similar in sites with annual prescribed burning (richness: 4; abundance: 44, Shannon diversity index: 1.0) compared to sites that had not been burned for three or 12 years (richness: 3–4; abundance: 22–23, Shannon diversity index: 0.8–1.0). Some lizard species (e.g. *Kentropyx viridistriga* and *Teius oculatus*) were more abundant in annually burned sites, whereas others (e.g. *Mabuya dorsivittata*) were more abundant in unburned sites (see original paper for details). One site each (≥ 400 ha) was burned annually (August–September), left unburned for three years or 12 years. Monitoring was undertaken using drift-fencing with pitfall traps in January–April 2006 (80 survey days).

A controlled study in 2005–2010 in a mixed coastal wetland, grass and scrubland and woodland habitat in California, USA (7) found that four years after prescribed burning, western yellow-bellied racer snake *Coluber constrictor mormon* abundance was lower in burned than unburned sites, but that abundance was similar in burned and unburned sites from five years after burning took place. Four years after prescribed burning, western yellow-bellied racer snake abundance was lower (2008: 17 snakes/trap array) compared to unburned sites

(49). In the fifth and sixth years after burning, snake abundance was similar in burned and unburned sites (2009 burned: 16 snakes/trap array vs. unburned: 25; 2010 burned: 19 vs. unburned: 30). Prescribed burns were carried out in a 213 ha area in autumn 2005 (64 ha) and 2006 (67 ha). Reptiles were surveyed in burned and adjacent unburned areas using traps, observation and coverboards. Traps were set in March–August 2007–2010 (277–1,140 trap days/year). Caught snakes (692 total individuals) were individually marked using PIT tags. Too few individuals were caught in the 2006 burn site to be included in analysis.

A controlled, before-and-after study in 2006 and 2010 in two abandoned agricultural fields in New York State, USA (8) found that prescribed burning increased the numbers of eastern massasauga rattlesnakes *Sistrurus catenatus catenatus* observed compared to before the fire and in an unburned area. After prescribed burning, eastern massasauga rattlesnakes were observed 27 times compared to no observations prior to burning and no observations in an unburned site over the same time period. The authors reported that rattlesnake occurrence was related to open habitats with low cover of leafy, non-woody plants (forbs) created by fire (see original paper for details). The study took place in two abandoned agricultural fields (disused for 15–20 years), one of which was burned in April 2010. Snakes were monitored using coverboards (in a 5 x 5 grid) per field before burning in 2006 and after burning in June–August 2010.

A replicated, controlled, before-and-after study in 2011–2012 in four riparian grasslands in Missouri, USA (9) found that areas with prescribed burning had higher reptile species richness compared to unburned areas. All results were reported as statistical model outputs. Reptile species richness was slightly higher in burned plots compared to unburned plots. Six turtles were found dead as a result of fire (two ornate box turtles *Terrapene ornata*, a western painted turtle *Chrysemys picta bellii* and three unidentified species). Snake presence was associated with 70–100% grass cover habitat that occurred the year following burning. Lizards were associated with burned or burned and heavily grazed plots, and turtles were associated with taller grass heights linked with light grazing. Patches of four watersheds (10–54 ha) were treated with prescribed burning (April 2011 or 2012) or were unmanaged during the past five years. Reptile monitoring took place 2–3 times/month in March–May 2011–2012 using coverboards and visual encounter surveys.

A controlled, before-and-after study in 2006 and 2010 in disused crop fields in New York State, USA (10) reported that following prescribed burning, the abundance of four snake species did not increase. Results were not statistically tested. Two months after a prescribed burn in a field, counts were similar for eastern milksnakes *Lampropeltis triangulum triangulum* (0.002 snakes/coverboard), northern brownsnakes *Storeria dekayi dekayi* (0.040), eastern gartersnakes *Thamnophis sirtalis sirtalis* (0.181) and northern watersnakes *Nerodia sipedon sipedon* (0.004) compared to four years earlier (in 2006 milksnake: 0.001 snakes/coverboard; brownsnake: 0.020; gartersnake: 0.230; watersnake 0). The authors reported that counts of eastern milksnakes, northern brownsnakes and eastern gartersnakes may have declined in a neighbouring field that wasn't burned over the same time period (see original paper for details). Snakes were monitored in two abandoned agricultural fields (1 km apart) that had been planted with crops until 15–20 years prior to the study, after which they had been managed by mowing biannually. Prescribed burning took place in one field in April 2010 instead of mowing in that year. Snakes were surveyed using coverboards in June-August 2006 and 2010 (25 coverboards/field, 20 total surveys).

A replicated, before-and-after study in 2008–2010 in savanna and open woodland in north Queensland, Australia (11) found that while overall reptile abundance reduced immediately after prescribed burns compared to pre-burn in three different grass types, there was no difference in reptile abundance compared to pre- or post-burn once sites had revegetated. Reptile abundance was lower immediately post-burn compared to pre-burn (average abundance post-burn: 1.2–3.6; pre-burn: 3.2–5.2 reptiles/grass type), but there was no difference in abundance levels after plots had revegetated compared to pre-burn or immediately post-burn (average abundance up to 15 months post burn: 2.1–4.1 reptiles/grass type). Reptile species composition differed between pre-burn and post-burn in all three grass types (data reported as statistical model results). Reptile species composition also differed between pre-burn and revegetated plots for one native grass type (kangaroo grass *Themeda triandra*), but not for the other two. Eight plots for each of three different grass types (kangaroo grass, black spear grass *Heteropogon contortus*, and non-native grader grass *Themeda quadrivalvis*) were monitored in 2008–2010 (24 plots in total). Monitoring was undertaken pre-burn (>2 years before last burn), immediately post-burn and following revegetation (up to 15 months post-burn) using drift fences with pitfall and funnel traps.

A replicated, randomized, controlled study in 2014–2015 in six rock and grassland areas in Australian Capital Territory, Australia (12) found that following prescribed burns of rocky outcrops, Australian pink-tailed worm-lizards *Aprasia parapulchella* recolonised some rock outcrops within one year. No statistical analyses were carried out. Two worm-lizards were observed on plots that were burned compared to zero on unburned plots. A further four worm-lizards were observed in nearby high-quality habitat (4 worm-lizards and 3 shed skins observed). In April–May 2014, plots (4 x 4 m) in six replicate sites (150 m apart) were each randomly selected and burned (using a blow torch; one plot/site) or left unburned (one plot/site). A further plot at each site of high-quality habitat was also monitored. In February 2015, rocks were surveyed for lizards. All sightings of worm-lizards or shed skins were recorded.

A site comparison study in 2016 in an area of heathland in Nouvelle-Aquitaine, France (13) found that one of six reptile species was less abundant in sites that were burned 5–12 years previously than in a site grazed by sheep, whereas the other five species were similarly abundant across all sites. Fewer western green lizards *Lacerta bilineata* were found in any of the burned sites (0.1 lizards/site for all burned sites) than in the grazed site (1.5 lizards/site), whereas no difference was found between burned or grazed sites in the number of wall lizards *Podarcis muralis* (0–4 lizards/site) or the number of four snakes species (green whip snake *Hierophis viridiflavus*, viperine snake *Natrix maura*, grass snake *Natrix natrix* and European asp *Vipera aspis*; data not presented). An area of heathland (135 ha) was managed by prescribed burning or annual sheep grazing. Three burned sites (one each burned 5, 10 or 12 years ago) and one grazed site (all sites 8–10 ha) were selected. In 2016, a total of 96 cover boards (corrugated roofing tiles) were split between the four areas (24 boards/area), and 10 surveys were conducted in April–June. Reptiles found on or under cover boards were counted.

A replicated, randomized, site comparison study in 2014–2016 in shrub and grass sandplain in Western Australia, Australia (14) found that small-scale patch burning was associated with increased sand goanna *Varanus gouldii* burrow abundance. Following several decades of annual small-scale burns, more sand goanna burrows were found in areas with a diverse burn history (results reported as statistical model outputs). Sand goanna burrows were particularly associated with no or early spinifex regrowth and mature, ready-to-be-burned spinifex habitats. The authors noted that sand goanna burrows found in plots with no or initial regrowth were likely to have been selected by over-wintering goannas prior to burning when the habitat was mature spinifex. Martu Aboriginal communities returned to the study area in 1984 and reinstated traditional winter patch burning since then. In July 2014–July 2016, seventy-six randomly-selected 1 ha plots (>1 km apart) in spinifex-dominated *Troidia* spp. desert were surveyed for sand goanna burrows. Plots were classified as: no regrowth present, early shoots present, mature plants with high plant diversity, mature spinifex able to carry a fire and deteriorating spinifex.

(1) Masters P. (1996) The effects of fire-driven succession on reptiles in spinifex grasslands at Uluru National Park, Northern Territory. *Wildlife Research*, 23, 39–47.

(2) Wilgers D.J. & Horne E.A. (2006) Effects of different burn regimes on tallgrass prairie herpetofaunal species diversity and community composition in the Flint Hills, Kansas. *Journal of Herpetology*, 40, 73–84.

(3) Masterson G.P., Maritz B. & Alexander G.J. (2008) Effect of fire history and vegetation structure on herpetofauna in a South African grassland. *Applied Herpetology*, 5, 129–143.

(4) Radke N.J., Wester D.B., Perry G. & Rideout-Hanzak S. (2008) Short-term effects of prescribed fire on lizards in mesquite-Ashe juniper vegetation in central Texas. *Applied Herpetology*, 5, 281–292.

(5) Ruthven D.C., Kazmaier R.T. & Janis M.W. (2008) Short-term response of herpetofauna to various burning regimes in the south Texas plains. *The Southwestern Naturalist*, *53*, 480–488.

(6) Cano P.D. & Leynaud G.C. (2010) Effects of fire and cattle grazing on amphibians and lizards in northeastern Argentina (Humid Chaco). *European Journal of Wildlife Research*, 56, 411–420.

(7) Thompson M.E., Halstead B.J., Wylie G.D., Amarello M., Smith J.J., Casazza M.L. & Routman E.J. (2013) Effects of prescribed fire on *Coluber constrictor mormon* in coastal San Mateo County, California. *Herpetological Conservation and Biology*, 8, 602–615.

(8) Dovčiak M., Osborne P.A., Patrick D.A. & Gibbs J.P. (2014) Conservation potential of prescribed fire for maintaining habitats and populations of an endangered rattlesnake *Sistrurus C. Catenatus*. *Endangered Species Research*, 22, 51–60.

(9) Larson D. (2014) Grassland fire and cattle grazing regulate reptile and amphibian assembly among patches. *Environmental Management*, 54, 1434–1444.

(10) Steen D.A., Osborne P.A., Dovčiak M., Patrick D.A. & Gibbs J.P. (2015) A preliminary investigation into the short-term effects of a prescribed fire on habitat quality for a snake assemblage. *Herpetological Conservation and Biology*, 10, 263–272.

(11) Abom R. & Schwarzkopf L. (2016) Short-term responses of reptile assemblages to fire in native and weedy tropical savannah. *Global Ecology and Conservation*, 6, 58–66.

(12) McDougall A., Milner R.N.C., Driscoll D.A. & Smith A.L. (2016) Restoration rocks: integrating abiotic and biotic habitat restoration to conserve threatened species and reduce fire fuel load. *Biodiversity and Conservation*, 25, 1529–1542.

(13) Pernat A., Sellier Y., Préau C. & Beaune D. (2017) Effet du pâturage sur le lézard vert occidental (*Lacerta bilineata* Daudin, 1802) (Squamata: Lacertidae) en milieu de landes. *Bulletin de la Société Herpétologique de France*, 161, 57–66.

(14) Bird R.B., Bird D.W., Fernandez L.E., Taylor N., Taylor W. & Nimmo D. (2018) Aboriginal burning promotes fine-scale pyrodiversity and native predators in Australia's Western Desert. *Biological Conservation*, 219, 110–118.

Wetland

- **Two studies** evaluated the effects of using prescribed burning in wetlands on reptile populations. One study was in each of the USA[1] and Australia[2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Abundance (2 studies):** One of two controlled studies (including one replicated, randomized study) in the USA[1] and Australia[2] found mixed effects of using prescribed burning in wetlands on the abundance of western yellow-bellied racer snakes[1]. The other study[2] found that found that burned areas had a similar abundance of reptiles and amphibians compared to unburned areas, but that delicate skinks were less abundant in burned areas.

BEHAVIOUR (0 STUDIES)

A controlled study in 2005–2010 in a mixed coastal wetland, grass and scrubland and woodland habitat in California, USA (1) found that four years after prescribed burning, western yellow-bellied racer snake *Coluber constrictor mormon* abundance was lower in burned than unburned sites, but that abundance was similar in burned and unburned sites from five years after burning took place. Four years after prescribed burning, western yellow-bellied racer snake abundance was lower (2008: 17 snakes/trap array) compared to unburned sites (49). In the fifth and sixth years after burning, snake abundance was similar in burned and unburned sites (2009 burned: 16 snakes/trap array vs. unburned: 25; 2010 burned: 19 vs. unburned: 30). Prescribed burns were carried out in a 213 ha area in autumn 2005 (64 ha) and 2006 (67 ha). Reptiles were surveyed in burned and adjacent unburned areas using traps, observation and coverboards. Traps were set in March–August 2007–2010 (277–1,140 trap days/year). Caught snakes (692 total individuals) were individually marked using PIT tags. Too few individuals were caught in the 2006 burn site to be included in analysis.

A replicated, randomized, controlled study in 2004–2006 in a seasonal wetland in Queensland, Australia (2) found that overall reptile and amphibian abundances were not affected by burning to remove invasive non-native para grass *Urochloa mutica,* but that the abundance of one skink species *Lampropholis delicata* was reduced in burned areas. When burns were carried out to control non-native para grass, overall reptile and amphibian abundance was similar to unburned plots (results presented as statistical model outputs) but abundance of *Lampropholis delicata* was lower in burned plots (3 skinks/plot) compared to unburned plots (14 skinks/plot). Para-grass dominated habitat in a conservation park (3,245 ha) was divided into plots (200 x 300 m each) that were either burned or unburned (3 plots/management type). Burning took place in August 2004, September 2005 and November 2006. Reptile and frog communities were sampled four times between 2005–2007 using three pitfall/funnel trap arrays/plot (see original paper for details). Reptiles were individually marked by toe clipping prior to release.

(1) Thompson M.E., Halstead B.J., Wylie G.D., Amarello M., Smith J.J., Casazza M.L. & Routman E.J. (2013) Effects of prescribed fire on *Coluber constrictor mormon* in coastal San Mateo County, California. *Herpetological Conservation and Biology*, 8, 602–615.
(2) Bower D.S., Valentine L.E., Grice A.C., Hodgson L. & Schwarzkopf L. (2014) A trade-off in conservation: Weed management decreases the abundance of common reptile and frog species while restoring an invaded floodplain. *Biological Conservation*, 179, 123–128.

8.2. Use prescribed burning in combination with vegetation cutting

- **Ten studies** evaluated the effects of using prescribed burning in combination with vegetation cutting on reptile populations. Eight studies were in the USA[1-3,5-8,10] and two were in Australia[4,9].

COMMUNITY RESPONSE (5 STUDIES)

- **Community composition (1 study):** One replicated, randomized, controlled, before-and-after study in the USA[6] found that cutting vegetation prior to burning resulted in reptile assemblages becoming similar to areas with more pristine habitat and a history of frequent fires.
- **Richness/diversity (5 studies):** Four of five replicated studies (including three randomized, controlled studies) in Australia[9,4] and the USA[2,5,10] found that areas managed by burning in combination with vegetation cutting had similar reptile species richness compared to either burning only[2,5,10], cutting only[2,5] or areas that were unmanaged[2,5,9,10]. The other study[4] found that areas of woodland managed by burning and vegetation thinning had higher reptile species richness than unmanaged areas.

POPULATION RESPONSE (9 STUDIES)

- **Abundance (9 studies):** Four of nine replicated studies (including five randomized, controlled studies) in the USA[1-3,5,7,8,10] and Australia[4,9] found that areas that were managed by burning in combination with vegetation cutting had a higher abundance of overall reptiles[3,4], lizards[3,5], eastern fence lizards[10] and five-lined skinks[10] compared to areas that were either only burned or unmanaged. Three studies[3,5,9] found a similar abundance of overall reptiles[5,9], snakes[3,5] and turtles[5] compared to either burning only, cutting only or unmanaged. Four studies[1,2,7,8] found mixed effects of burning in combination with vegetation cutting on the abundance of reptiles[1,2,8] and six-lined racerunners[7].

BEHAVIOUR (0 STUDIES)

Background
Prescribed burning may be used to reduce the chance of more extensive and damaging wildfires and to maintain and restore habitats historically subject to occasional wildfires. Using prescribed burning alongside mechanical cutting and clearing of vegetation may combine the multiple ecosystem functions provided by fire with the increased selectivity of cutting.

For studies that assess these actions separately see *Use prescribed burning* and *Habitat restoration and creation – Manage vegetation by cutting or mowing*.

A replicated, randomized, controlled study in 1997–1998 of pine sandhills in Florida, USA (1) found that using prescribed burning in combination with tree felling/girdling had mixed effects depending on species and year. In one of two burn years, capture rates of six-lined racerunners *Cnemidophorus sexlineatus* and eastern fence lizards *Sceloporus undulatus* were lower in plots with burning and tree felling (racerunners: 0.031 captures/trap days; fence lizards: 0.003) than in burn-only plots (racerunners: 0.037; fence lizards: 0.007), but were higher than in plots with no cutting or burning (racerunners: 0.015; fence lizards: 0.002). Southeastern crowned snake *Tantilla coronata* capture rates were similar in felling and burning plots compared to burn only plots in both years (0.004–0.007 and 0.011–0.024), but lower than plots with no burning or felling in one year (felling and burning: 0.007; no felling or burning: 0.014) but not the other (felling and burning: 0.024; no felling or burning: 0.019). Green anoles *Anolis carolinensis* were not caught in burning and felling plots and were caught a similar amount in burn only and no burning or felling plots (0.003). Little brown skink *Scincella lateralis* captures were similar across all treatments (0.002–0.005). Treatments (burning with tree felling/girdling or burn only) were randomly assigned to 81 ha plots within four replicate blocks. Burn-only and felling/girdling treatments were carried out in spring 1995. Felling/girdling plots were subsequently burned in March–April 1997. Monitoring was undertaken using drift-fencing and pitfall traps in April–August 1997–1998.

A replicated, randomized, controlled, before-and-after study in 2001–2004 in an upland hardwood forest in North Carolina, USA (2, same experimental set-up as 10) found that using prescribed burning combined with mechanically removing understory vegetation did not increase reptile species richness, but had mixed effects on reptile abundance depending on the species or species group. In the two years after management was carried out, reptile species richness was similar in plots with burning and mechanical vegetation removal, burning only and plots with no management (1–4 species/plot). Total reptile abundance was higher with burning and mechanical vegetation removal (6.0–8.7 reptiles/100 nights) compared to burning alone (1.4–2.7), but no different from either mechanical vegetation removal only (4.4–4.5) or unmanaged (3.0–3.2). Eastern fence lizards *Sceloporus undulatus* were also more abundant with burning and mechanical vegetation removal (1.9–2.7 individuals/100 nights) compared to burning alone (0–0.2), but similar to mechanical removal only (0.5–1.4) and no management (0.5–0.8). See original paper for other individual species abundances. Three forest segments were divided into four management zones (14 ha each): prescribed burn with mechanical vegetation removal, prescribed burn only, mechanical vegetation removal only, and no management. Chainsaws were used to remove mid-storey vegetation in winter 2001–2002 and prescribed burns took place in March 2003. Reptiles were surveyed using drift fence arrays with pitfall and funnel traps before any management took place in August–October 2001 and after management in May–September 2002–2004.

A replicated, site comparison study in 1999–2001 of pine woodland in western Arkansas, USA (3) found that restoring woodland by controlled burning and thinning trees resulted in higher abundance of reptiles compared to unrestored woodland. Overall reptile captures were higher in plots with burning and thinning (78 lizards/plot) than in unmanaged plots (54 lizards/plot); as were overall lizard captures (42 vs 28); whereas no difference was found for snakes (34 vs 25) or three-toed box turtles *Terrapene carolina triunguis* (1 vs 1; the only turtle species caught). Nine plots (11–42 ha) that had been thinned (1980–1990) and then burned at least three times at 3–5-year intervals were sampled. These were compared to three unmanaged, unburned plots. Controlled burns were in March–April. Three drift-fence arrays with pitfall and box traps were established/plot. Traps were checked weekly in April-September 1999–2001.

A replicated, controlled study in 2002–2006 of forest at a site in Western Australia, Australia (4) found that burning vegetation and thinning trees, as part of post-mining restoration, increased reptile abundance and species richness. Reptile abundance and richness in thinned and burned plots (abundance: 7–8 individuals/grid, richness: 4 species/grid) was higher than in plots that were not thinned and burned (abundance: 4–5 individuals/grid, richness: 2 species/grid). See paper for details of individual species. In 1984–1992, areas of a former bauxite mine were either planted with non-local tree species or sown with the seed of local tree species. Eight plots were thinned between December 2002 and July 2003 and then burned in November 2003. An additional eight plots were not thinned or burned. Reptiles were monitored for four nights each in October and November–December 2005 and March and May 2006, using pitfall traps with drift fencing and live cage and box traps.

A replicated, randomized, controlled study in 2006–2007 in hardwood forests in North Carolina, USA (5) found overall reptile species richness and capture rates were similar after burning combined with mechanical vegetation cutting or burning or cutting alone, but that lizard capture rates were mostly higher after mechanical-cutting with burning compared to other management options. Overall reptile richness and overall reptile, snake and turtle captures were similar after burning with mechanical-cutting (richness: 5–8 species/100 array nights, overall captures: 12–13 individuals/100 array nights; snakes: 2 individuals/100 array nights; turtles: 0 individuals/100 array nights), twice-burning (4–7, 7–9, 2–7, 0–1), mechanical-cutting only (6–7, 6, 1–2, 0), and no management (6, 7–7, 3–5, 0). Lizard captures were higher after burning with mechanical-cutting (11 individuals/100 array nights) compared to twice-burning (3–5) or no management (4). In the first monitoring year, lizard captures were higher after burning with mechanical-cutting than mechanical-cutting only (4 individuals/100 array nights) but were statistically similar in the second year of monitoring (burning with cutting: 11; cutting only: 5). Three blocks of four sets of 10 ha sites were managed with mechanical-cutting followed by twice-burning, mechanical-cutting (using chainsaws to cut trees and understory, 2001–2002), twice-burning (in March 2003 and February 2006) or no management. Reptiles were surveyed in May – August 2006 and 2007 using drift fences with pitfall traps ('arrays', 3/site).

A replicated, randomized, controlled, before-and-after study in 1995–2010 in fire-suppressed longleaf pine *Pinus palustris* forest in Florida, USA (6, same experimental set-up as 7) found that cutting vegetation prior to burning resulted in reptile assemblages becoming similar to both unburned areas and areas of more pristine habitat. All results reported as statistical model outputs, see original paper for details. Reptile communities in sites treated by cutting vegetation followed by burning were similar to unburned sites and areas of more pristine habitat, but composition was initially different to burn only areas. After a further 10–12 years of all sites receiving regular burning, all reptile communities became similar to areas of pristine habitat. See original paper for details of individual species responses to management. Reptiles were monitored in four sites each (81 ha each) managed by: vegetation cutting (felling and girdling in June–November 1995, 4 sites) with burning (1997), prescribed burning (April–June 1995, 4 sites), or unmanaged until after 1999 when all sites were burned at 2–3 year intervals. Reptiles were also monitored a further four sites in an area without historical fire suppression. Reptiles were surveyed using drift fences with pitfall traps (16 traps/site) in April–August 1997–1998 and May–September 2009–2010.

A replicated, randomized, controlled, before-and-after study in 1995–2010 in fire-suppressed longleaf pine *Pinus palustris* forests in Florida, USA (7, same experimental set-up as 6) found that areas where vegetation was cut followed by burning had similar numbers of six-lined racerunners *Aspidoscelis sextineatus* compared to burn only and pristine habitat areas, whereas unburned areas had fewer than pristine areas. Areas with vegetation cutting followed by burning had similar numbers of six-lined racerunner (adults: 24; juveniles: 5) as burn only areas (23, 10) and areas of more pristine habitat (38, 10), whereas unburned areas had fewer than pristine areas (13, 2). After a further 10–12 years of prescribed burns on all sites, six-lined racerunner abundances were similar in sites managed initially by vegetation cutting and burning (adults: 28 individuals/site; juveniles: 10), burning (40. 7), unburned sites (30, 6) and areas of pristine habitat (37, 10). Reptiles were monitored in six sites each (81 ha each) managed by: vegetation cutting (felling and girdling in June–November 1995, 6 sites) with burning (1997), prescribed burning (April–June 1995, 6 sites) or unmanaged until after 1999 when all sites were burned at 2–3-year intervals. Reptiles were also monitored at a further six sites in an area without historical fire suppression. Reptiles were surveyed using drift fences with pitfall traps (16 traps/site) in April–August 1997–1998 and May–September 2009–2010.

A replicated, controlled, before-and-after study in 2005–2008 in mixed forest in Alabama, USA (8) found that the effect of prescribed burning with thinning trees or burning alone was mixed depending on reptile species. Eastern fence lizard *Sceloporus undulatus* captures increased after burning in burn-only stands (pre-burn: 0 individuals/100 trap nights, post-burn 1–4) and were higher in the second year after heavy thinning with burning (13) compared to no management (1). Green anole *Anolis carolinensis* captures were higher in the first year after thinning with burning (17–18) and thinning (13) compared to burn only (0), but similar to no management (5). Little brown skink *Scincella lateralis* captures decreased in the first year after all management (2–3 individuals/100 trap nights) compared to pre-management (4–13). Five-lined skink *Plestiodon fasciatus* captures were lower in the first year after burning (0) compared to the first year after heavy thinning (7). See paper for details of other species responses. In 2005–2008, the impact of six management options (burn only, light tree thinning, heavy thinning, light thinning with burning, heavy thinning with burning and no management) on reptiles were tested in three blocks of six 9 ha plots. Reptiles were surveyed for 3–

6 months before management began (564 total trap nights in April–August) and in the two years after management (3,132 total trap nights in March–September) using drift fences with pitfall traps.

A replicated, site comparison study in 1990–1992, 2005–2006 and 2010–2011 in eucalypt forest in Western Australia, Australia (9) found that burned and thinned restored ex-mining forest had similar species richness and abundance to unmanaged restored ex-mining forest, but restored forest overall had lower species richness compared to unmined forest. Seven years after 20–22-year-old restored mining forest was managed through prescribed burning and tree thinning, reptile species richness was similar between managed-restored forest (5 species/plot) and unmanaged-restored forest (4) but richness in both was lower than in unmined forest (9). Reptile abundance was statistically similar in managed-restored (21 individuals/plot) and unmanaged-restored forest (10) and unmanaged-restored forest had lower abundance than unmined forest (34). See original paper for individual reptile abundances. The area was restored after mining in 1990–1992 by reseeding with local over- and understory species. Reptiles were surveyed in four plots of each of managed-restored, unmanaged-restored, and unmined forest. Managed-restored forest was thinned by felling trees (December 2002–June 2003) and prescribed burning (November 2003, reduced to 600–800 stems/ha) and two plots were re-thinned in January–December 2009 (reduced to 400 stems/ha). Unmined forest was prescribed burned 3–5 years before surveys. Reptiles were monitored using drift fences with funnel and pitfall traps in 2005–2006, 2010, and 2011.

A replicated, randomized, controlled study in 2001–2016 in upland forest in North Carolina, USA (10, same experimental set-up as 2) found that prescribed burning combined with mechanical understory removal did not increase overall species richness compared to burning alone or no management, but that two lizard species abundances were greater after prescribed burning with mechanical removal compared to no management. Overall reptile species richness was similar between prescribed burning with mechanical removal, prescribed burning only, mechanical removal only, and unmanaged forest (data reported as model outputs). In 2016, five-lined skink *Plestiodon fasciatus* and eastern fence lizard *Sceloporus undulatus* capture rates were greater after burning with mechanical removal, but not burning or mechanical removal alone, than in unmanaged forest (skink - burning with mechanical: 4 skinks/100 array nights, burning only: 2.8, mechanical only: 1.6, no management: 1.6; lizard - burning plus mechanical: 5.3 lizards/100 array nights, burning only: 3.6, mechanical only: 2.4, no management: 0.5). Three study sites were selected within a 5,841 ha mixed oak-hickory forest. Within each site, experimental plots (10 ha core areas with 20 m wide buffers) were managed as follows: prescribed burning only (2003, 2006, 2012, 2015); mechanical understory removal (in winters 2001–2002 and 2011–2012); mechanical understory removal (winter 2001–2002) followed by prescribed burning (2003, 2006, 2012, 2015) and unmanaged. Reptiles were surveyed after management using drift fences with pitfall and funnel traps ('arrays') in May – August of 2003–2004, 2006–2007, 2014 and 2015–2016 (158–341 array nights/plot/year).

(1) Litt A.R., Provencher L., Tanner G.W. & Franz R. (2001) Herpetofaunal responses to restoration treatments of longleaf pine sandhills in Florida. *Restoration Ecology*, 9, 462–474.

(2) Greenberg C.H. & Waldrop T.A. (2008) Short-term response of reptiles and amphibians to prescribed fire and mechanical fuel reduction in a southern Appalachian upland hardwood forest. *Forest Ecology and Management*, 255, 2883–2893.
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(6) Steen D.A., Smith L.L., Conner L.M., Litt A.R., Provencher L., Hiers J.K., Pokswinksi S. & Guyer C. (2013) Reptile assemblage response to restoration of fire-suppressed longleaf pine sandhills. *Ecological Applications*, 23, 148–158.
(7) Steen D.A., Smith L.L., Morris G., Conner L.M., Litt A.R., Pokswinski S. & Guyer C. (2013) Response of six-lined racerunner (*Aspidoscelis sexlineata*) to habitat restoration in fire-suppressed longleaf pine (*Pinus palustris*) sandhills. *Restoration Ecology*, 21, 457–463.
(8) Sutton W.B., Wang Y. & Schweitzer C.J. (2013) Amphibian and reptile responses to thinning and prescribed burning in mixed pine-hardwood forests of northwestern Alabama, USA. *Forest Ecology and Management*, 295, 213–227.
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(10) Greenberg C.H., Moorman C.E., Matthews-Snoberger C.E., Waldrop T.A., Simon D., Heh A. & Hagan D. (2018) Long-term herpetofaunal response to repeated fuel reduction treatments. *Journal of Wildlife Management*, 82, 553–565.

8.3. Use prescribed burning in combination with herbicide application

- **Five studies** evaluated the effects of using prescribed burning in combination with herbicide application on reptile populations. Four studies were in the USA[1-4] and one was in Australia[5].

COMMUNITY RESPONSE (2 STUDIES)

- **Community composition (1 study):** One replicated, randomized, controlled, before-and-after study in the USA[2] found that reptile community composition responded differently to herbicide treatment followed by burning or burning alone when compared to unburned areas or areas of more pristine habitat.
- **Richness/diversity (1 study):** One replicated, randomized, controlled, before-and-after study in the USA[4] found that areas that were burned in combination with herbicide application had similar combined reptile and amphibian species richness and diversity compared to areas that were managed by burning or herbicide application alone or left unmanaged.

POPULATION RESPONSE (3 STUDIES)

- **Abundance (3 studies):** Two of three replicated, randomized, controlled studies (including two before-and-after studies) in the USA[1,3,4] found mixed effects of burning in combination with herbicide application on the abundance of reptiles[1] and six-lined racerunners[3]. The other study[4] found that areas that were burned in combination with herbicide application had a similar abundance of reptiles compared to areas that were managed by burning or herbicide application alone or left unmanaged. The study[4] also found that the abundance of eastern fence lizards was higher in the first year after burning and herbicide application compared to unmanaged areas, but similar for the next six years.

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One replicated, randomized, controlled study in Australia[5] found that some rocky outcrops that were burned in combination with herbicide application were recolonized by pink-tailed worm-lizards.

Background

Prescribed burning may be used to reduce the chance of more extensive and damaging wildfires and to maintain and restore habitats historically subject to occasional wildfires. Using prescribed burning alongside herbicide application may combine the multiple ecosystem functions provided by fire with the increased selectivity of herbicides.

For studies that assess these actions separately see *Use prescribed burning* and *Habitat restoration and creation – Manage vegetation using herbicides*.

A replicated, randomized, controlled study in 1997–1998 of pine sandhills in Florida, USA (1) found that burning in combination with herbicide had mixed effects depending on species and year. In one of two burn years, capture rates of six-lined racerunners *Cnemidophorus sexlineatus* and eastern fence lizards *Sceloporus undulatus* were lower in plots with burning and herbicide (six-lined racerunners: 0.016 captures/trap days; eastern fence lizards: 0.003) than in burn-only plots (racerunners: 0.037; fence lizards: 0.007), but higher than in unburned plots with no herbicide (racerunners: 0.015; fence lizards: 0.002). In the other year, captures were similar all three treatments (racerunners: 0.011–0.017; fence lizards: 0.001–0.006). In one of two years southeastern crowned snake *Tantilla coronata* captures were lower in burning with herbicide and burning only plots (0.004–0.005) compared to plots with no burning or herbicide (0.014), but in the other year captures were similar (0.011–0.020). Green anoles *Anolis carolinensis* were not caught in burning and herbicide plots and were caught a similar amount in burn only and no burning or herbicide plots (0.003). Little brown skink *Scincella lateralis* captures were similar across all treatments (0.001–0.005). Treatments (burning with herbicide or burn only) were randomly assigned to 81 ha plots within four replicate blocks. Burn-only treatments were carried out in spring 1995. Herbicide treatments were carried out in 1995 and were then burned in March–April 1997. Data were also collected from four frequently-burned reference sites. Monitoring was undertaken using drift-fencing and pitfall traps in April–August 1997–1998.

A replicated, randomized, controlled, before-and-after study in 1995–2010 in fire-suppressed longleaf pine *Pinus palustris* forest in Florida, USA (2, same experimental set-up as 3) found reptile community composition responded differently to herbicide treatment followed by burning or to burning alone when compared to unburned areas. All results reported as statistical model outputs, see original paper for details. Reptile communities in sites treated with herbicide then burned remained similar to unburned sites and different to areas of pristine habitat (1–2 years post treatment). Reptile communities in sites that were only burned were different to both unburned sites and areas of pristine habitat. After

10–12 years of all sites receiving regular burning, all reptile communities became similar to areas of pristine habitat. See original paper for details of individual species responses to management. Reptiles were monitored in four sites (81 ha each) each managed by: prescribed burning (April–June 1995, 4 sites), using herbicides (May 1995, 4 sites) with burning (1997), or unburned and no herbicide until after 1999 when all sites were burned at 2–3-year intervals. Reptiles were also monitored a further four sites in an area of pristine habitat with a history of regular fires. Reptiles were surveyed using drift fences with pitfall traps (16 traps/site) in April–August 1997–1998 and May–September 2009–2010.

A replicated, randomized, controlled, before-and-after study in 1995–2010 in fire-suppressed longleaf pine *Pinus palustris* forests in Florida, USA (3, same experimental set-up as 2) found that areas where herbicide was applied prior to regular burning had similar numbers of six-lined racerunners *Aspidoscelis sexlineatus* compared to unburned areas and fewer than areas of pristine habitat, whereas burn only sites had similar numbers to areas of pristine habitat. Six-lined racerunner abundances were lower in sites treated with herbicide followed by burning (adults: 14 individuals/site; juveniles: 4) and unburned and no herbicide sites (13, 2) compared to burn-only sites (23, 10) or more pristine areas with a history of fires (38, 10). After a further 10–12 years of prescribed burns on all sites, six-lined racerunner abundances were similar in sites managed initially by burning (adult: 40 individuals/site; juvenile: 7), herbicide followed by burning (33, 6) unburned and no herbicide (30, 6) and pristine sites with a history of fires (37, 10). Reptiles were monitored in six sites each (81 ha each) managed by: prescribed burning (April–June 1995, 6 sites), herbicides (May 1995, 6 sites) followed by burning (1997), or unburned with no herbicide until after 1999 when all sites were burned at 2–3-year intervals. Reptiles were also monitored at a further six sites in a more pristine area with a history of fires. Reptiles were surveyed using drift fences with pitfall traps (16 traps/site) in April–August 1997–1998 and May– September 2009–2010.

A replicated, randomized, controlled, before-and-after study in 1999–2007 in six pine plantations in Mississippi, USA (4) found that prescribed burning in combination with herbicide application did not increase reptile and amphibian richness, diversity or most species abundances compared to burning or herbicide application alone or no management, though eastern fence lizard *Sceloporus undulatus* abundance was higher in the year after management for all treatment types. In six of seven years after burning and/or herbicide applications, species richness, diversity measures and most species abundances were similar in burn with herbicide, burn only, herbicide only and unmanaged plots (data reported as model outputs, see paper for details). Eastern fence lizard abundance was higher in managed plots (burn with herbicide applied, burn only and herbicide only: 0.02 lizards/plot) in the first year after management compared to unmanaged plots (0.002 lizards/plot). Four 10 ha plots were set up in six intensively-managed, 18–22-year-old commercial pine stands (59–120 ha). Plots were either burned in the dormant season (December–February) in 2000, 2003 and 2006 and treated with herbicide ('Imazapyr') in September 1999; burned only; treated with herbicide only; or unmanaged. Reptiles were monitored using drift fences with pitfall and funnel traps in May–June 1999–2007 (one year before management and seven years after management began).

A replicated, randomized, controlled study in 2014–2015 in six rock and grassland areas in Australian Capital Territory, Australia (5) found that after rocky outcrops were treated with prescribed burns in combination with herbicide application, some were recolonised by Australian pink-tailed worm-lizards *Aprasia parapulchella* within one year. Results were not statistically tested. Four worm-lizards were observed on plots treated with burning and herbicide, two on burn only plots, and zero were observed on unrestored plots. A further four worm-lizards were observed in nearby high-quality habitat (4 worm-lizards and 3 shed skins observed). In April–May 2014, plots (4 x 4 m) in six replicate sites (150 m apart) were each randomly assigned either burning and herbicide application (burned using a blow torch; one plot/site), burning only (one plot/site) or unburned (two plots/site, one adjacent to managed plots and the second in nearby high-quality lizard habitat). In February 2015, rocks were surveyed for lizards. All sightings or shed skins were recorded.

(1) Litt A.R., Provencher L., Tanner G.W. & Franz R. (2001) Herpetofaunal responses to restoration treatments of longleaf pine sandhills in Florida. *Restoration Ecology*, 9, 462–474.
(2) Steen D.A., Smith L.L., Conner L.M., Litt A.R., Provencher L., Hiers J.K., Pokswinksi S. & Guyer C. (2013) Reptile assemblage response to restoration of fire-suppressed longleaf pine sandhills. *Ecological Applications*, 23, 148–158.
(3) Steen D.A., Smith L.L., Morris G., Conner L.M., Litt A.R., Pokswinski S. & Guyer C. (2013) Response of six-lined racerunner (*Aspidoscelis sexlineata*) to habitat restoration in fire-suppressed longleaf pine (*Pinus palustris*) sandhills. *Restoration Ecology*, 21, 457–463.
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8.4. Use prescribed burning in combination with grazing

- **Five studies** evaluated the effects of using prescribed burning in combination with grazing on reptile populations. Two studies were in the USA[1,5], two were in Australia[2,4] and one was in Argentina[3].

COMMUNITY RESPONSE (3 STUDIES)

- **Richness/diversity (3 studies):** One of two studies (including one site comparison study and one replicated, controlled, before-and-after study) in Argentina[3] and the USA[5] found that areas that were burned in combination with grazing had similar reptile species richness and diversity compared to areas not burned or grazed for 3–12 years[3]. The other study[5] found that areas that were burned in combination with grazing had higher species richness than lightly grazed or unmanaged areas and similar richness compared to areas that were burned only. One before-and-after study in the USA[1] found that an area with annual prescribed burning combined with intensive early-season grazing had similar reptile species richness compared to when it was managed by alternate year prescribed burning with season-long grazing.

POPULATION RESPONSE (3 STUDIES)

- **Abundance (3 studies):** Two site comparison studies (including one replicated study) in Australia[2] and Argentina[3] found that that burning in combination with grazing had mixed effects on the abundance of reptile species. One replicated, randomized, controlled study in Australia[4] found that areas where invasive para grass was removed by burning in combination with grazing had similar overall reptile and amphibian abundance compared to areas that were only burned or unmanaged. The study[4] also found that the abundance of delicate skinks was lower in areas that were burned and grazed compared to those that were unmanaged.

BEHAVIOUR (0 STUDIES)

Background

Prescribed burning may be used to reduce the chance of more extensive and damaging wildfires and to maintain and restore habitats historically subject to occasional wildfires. Using prescribed burning alongside grazing may further alter vegetation height and cover as well as the abundance of certain plants and the diversity of plant communities.

For studies that assess these actions separately see *Use prescribed burning* and *Habitat restoration and creation – Manage vegetation using livestock grazing*.

A before-and-after study in 1989–2003 at a rangeland cattle ranch in Kansas, USA (1) found that annual prescribed burning combined with intensive early-season grazing resulted in similar reptile species richness compared to alternate year prescribed burning with season-long stocking. Six years after an alternate-year burning combined with season-long stocking regime began, species richness was estimated to be similar (32 species) compared to five years after the start of annual burning combined with intensive early cattle stocking (27 species). Four turtle species, six lizard species and 17 snake species were observed during the study. The authors reported that species loss rates were estimated to be higher following burn years (see paper for details). In 1989–1998, season-long stocking (200 cows with calves, 0.6 animals/ha) was combined with alternate year prescribed burning. On the same site in 1999–2003, intensive-early cattle stocking (650 yearling cattle, 1 animal/ha for 3 months starting in late spring) was combined with annual prescribed burning. In 1989–2003, visual surveys for reptiles were conducted on one day in mid-spring each year along a 4 km transect by turning over rocks and other debris and sighting animals in the open.

A replicated, site comparison study in 2001 in savanna woodlands in Queensland, Australia (2) found that overall reptile abundance was similar in burned and unburned areas regardless of grazing practices, though the abundance of one of 18 species was higher after burning and of another was lower after burning with grazing. Overall reptile abundance was similar in burned (12–20 individuals/plot) and unburned plots (14–19), regardless of grazing practices. Of 18 species recorded, one dragon species abundance was higher in burned than unburned plots regardless of grazing (central netted dragon *Ctenophorus nuchalis* burned: 0.7–1.0 individuals/plot; unburned: 0–0.1) and one ctenotus abundance was lower in burned than unburned plots, particularly when burning was combined with grazing (leopard ctenotus *Ctenotus pantherinus* burned: 0–1.4; unburned: 1.3–4.4). In January 2001, reptiles were monitored on three cattle stations (>20,000 ha each) in 29 one-ha plots that were either grazed (4–8 cattle/ha) or ungrazed (paddocks where cattle were excluded) and either recently burned (within 2 years) or unburned (last burnt >2 years ago). Burns were a mixture of prescribed burns and wildfires and all treatments took place over >2,000 ha areas. Reptiles were sampled using cage traps and pitfalls supplemented by day and night log rolling and litter raking.

A site comparison study in 2006 of cattle pasture in Corrientes, Argentina (3) found that overall reptile diversity, species richness and abundance were not significantly different following annual prescribed burning with or without livestock grazing. Overall reptile species richness, abundance and diversity were similar in sites with annual prescribed burning with or without grazing (richness: 4; abundance: 17–44, Shannon diversity index: 1.0–1.1) compared to sites that had not been burned or grazed for three or 12 years (richness: 3–4; abundance: 22–23, Shannon diversity index: 0.8–1.0). Some lizard species (e.g. *Kentropyx viridistriga* and *Teius oculatus*) were more abundant in annually burned sites, whereas others (e.g. *Mabuya dorsivittata*) were more abundant in unburned and ungrazed sites (see original paper for details). The four historical treatments (≥ 400 ha) were: annual prescribed burning (August–September) with or without grazing (3 ha/cattle unit), three years since a prescribed burning, and no fire or grazing for 12 years. Monitoring was undertaken using drift-fencing with pitfall traps in January–April 2006 (80 survey days).

A replicated, randomized, controlled study in 2004–2006 in a seasonal wetland in Queensland, Australia (4) found that overall reptile and amphibian abundance was not affected by burning, or burning and grazing to remove invasive non-native para grass *Urochloa mutica,* but that the abundance of one skink species *Lampropholis delicata* was reduced in burned and grazed and burn only plots. Overall reptile and amphibian abundance was similar in burned and grazed, burned and unmanaged plots (results presented as statistical model outputs). However, abundance of *Lampropholis delicata* was lower in all managed plots (burned: 3 skinks/plot; grazed: 4 skinks/plot; burned and grazed: 1 skinks/plot) compared to unmanaged plots (14 skinks/plot). Para-grass dominated habitat in a conservation park (3,245 ha) was divided into 12 plots (200 x 300 m each) and each plot was either burned, grazed, burned and grazed, or not managed (3 plots/management type). Burning took place in August 2004, September 2005 and November 2006. Cattle *Bos indicus* grazing took place after burning in September–December 2004, October–December 2005 and November–December 2006. Stocking levels were calculated to consume 50% of the grass biomass present/plot. Reptile and frog communities were sampled four times between 2005–2007 using three pitfall/funnel trap arrays/plot (see original paper for details). Reptiles were individually marked by toe clipping prior to release.

A replicated, controlled, before-and-after study in 2011–2012 in four riparian grasslands in Missouri, USA (5) found that prescribed burning with heavy grazing and burning alone increased reptile species richness compared to no management. All results were reported as statistical model outputs. Reptile species richness was slightly higher in burned and grazed or burned plots compared to unmanaged or lightly grazed plots. Six turtles were found dead as a result of fire (two ornate box turtles *Terrapene ornata*, a western painted turtle *Chrysemys picta bellii* and three unidentified species). Snake presence was associated with 70–100% grass cover habitat that occurred the year following burning, lizards were associated with burned or burned and heavily grazed plots, and turtles were associated with taller grass heights linked with light grazing.

Patches of four watersheds (10–54 ha) were treated with combinations of prescribed burning alone (April 2011 or 2012), or light grazing (May–July 2011 or 2012), or burning followed by heavy grazing (May–July after April burning in 2011 or 2012), or unmanaged during the past five years. Reptile monitoring took place 2–3 times/month in March–May 2011–2012 using coverboards and visual encounter surveys.

(1) Wilgers D.J., Horne E.A., Sandercock B.K. & Wolkmann A.W. (2006) Effects of rangeland management on community dynamics of the herpetofauna of the tallgrass prairie. *Herpetologica*, 62, 378–388.
(2) Kutt A.S. & Woinarski J.C.Z. (2007) The effects of grazing and fire on vegetation and the vertebrate assemblage in a tropical savannah woodland in north-eastern Australia. *Journal of Tropical Ecology*, 23, 95–106.
(3) Cano P.D. & Leynaud G.C. (2010) Effects of fire and cattle grazing on amphibians and lizards in northeastern Argentina (Humid Chaco). *European Journal of Wildlife Research*, 56, 411–420.
(4) Bower D.S., Valentine L.E., Grice A.C., Hodgson L. & Schwarzkopf L. (2014) A trade-off in conservation: Weed management decreases the abundance of common reptile and frog species while restoring an invaded floodplain. *Biological Conservation*, 179, 123–128.
(5) Larson D. (2014) Grassland fire and cattle grazing regulate reptile and amphibian assembly among patches. *Environmental Management*, 54, 1434–1444.

8.5. Create fire breaks

- **One study** evaluated the effects of creating fire breaks on reptile populations. This study was in Australia[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Abundance (1 study):** One replicated, controlled, before-and-after study in Australia[1] found that in areas with fire suppression measures combined with fences to exclude predators, reptile abundance increased over time.

BEHAVIOUR (0 STUDIES)

Background
In some environments, fires can damage important habitats, particularly if habitat patches are small or fragmented, meaning that entire patches can be destroyed in fires. Fire breaks are ploughed, open or unplanted gaps of land around the perimeters of, or spaced within, areas of forest, grassland or farmland intended to prevent the spread of fire, thereby protecting important habitats. While the primary purpose of creating firebreaks may be top prevent the spread of fire, the open areas created could act as suitable habitat for some species, or present barriers to the movement of others.

See also: *Put out wildfires*.

A replicated, controlled, before-and-after study in 2013–2015 in tropical savanna in the Northern Territory, Australia (1) found that reptile abundance remained similar in plots with fire breaks and active fire suppression compared to those with no breaks or suppression, though in fenced plots with fire breaks and suppression reptile abundance increased over time. Reptile abundance remained similar in plots with and without fire breaks (and fire suppression) that were also unfenced (2013: 0.6 reptiles/plot; 2015: 0.5 reptiles/plot; results standardised by sampling effort). In fenced areas, which all had fire breaks and suppression, average reptile abundance doubled over two years (2013: 0.3 reptiles/plot; 2015: 0.7 reptiles/plot; results standardised by sampling effort). Across all plots, reptile abundance increased with time since the last fire (0 months: 2 reptiles/plot; 50 months: 3 reptiles/plot). The effects of fire breaks and suppression and/or fencing on species richness was inconclusive (see original paper for details). Data were collected from six 64 ha plots, with two each treated with: fire breaks and suppression and no exclusion fencing, fire breaks and suppression and exclusion fencing; and no fire breaks, fire suppression or exclusion fencing. Fire breaks (8 m wide) were established around plot perimeters, and fuel reduction burning in the early dry season also took place, along with active fire suppression inside the plots (details not provided). Exclusion fences were installed in December 2013 (1,800 mm high and 550 mm below ground). Reptiles were monitored seasonally (March–April, June–July, October–November) in six transects/plot using drift fences with pitfall traps in 2013–2015.

(1) Stokeld D., Fisher A., Gentles T., Hill B.M., Woinarski J.C., Young S. & Gillespie G.R. (2018) Rapid increase of Australian tropical savanna reptile abundance following exclusion of feral cats. *Biological Conservation*, 225, 213–221.

8.6. Put out wildfires

- We found no studies that evaluated the effects of putting out wildfires on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

In some environments, fires can damage important habitats, particularly if habitat patches are small or fragmented, meaning that entire patches can be destroyed in fires. The impacts of large-scale wildfires may result in significant changes to species diversity and community composition in those areas that are affected (Rochester *et al*. 2010), though the effects of fires and differing fire histories on reptile species, species richness and community composition can be variable and difficult to predict (Lindenmayer *et al*. 2008 and references therein).

Testing this intervention may present a number of challenges, not least the potential need to allow some wildfires to burn that would have otherwise been put out. Practitioners should therefore carefully consider the harms that could be caused to people's health and livelihoods by allowing wildfires to burn, as well as the harms to other species and habitats.

See also: *Create fire breaks*

Lindenmayer D.B., Wood J.T., MacGregor C., Michael D.R., Cunningham R.B., Crane M., Montague-Drake R., Brown D., Muntz R. & Driscoll D.A. (2008) How predictable are reptile responses to wildfire? *Oikos*, 117, 1086–1097.

Rochester C.J., Brehme C.S., Clark D.R., Stokes D.C., Hathaway S.A. & Fisher R.N. (2010) Reptile and amphibian responses to large-scale wildfires in southern California. *Journal of Herpetology*, *44*, 333–351.

Water management and use

8.7. Regulate water levels

- **One study** evaluated the effects of regulating water levels on reptile populations. This study was in France[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Abundance (1 study):** One controlled, before-and-after study in France[1] found that autumn–spring marsh flooding with moderate levels of grazing in autumn–winter led to higher numbers of European pond turtles than winter–spring flooding with high levels of grazing in spring–summer.

BEHAVIOUR (0 STUDIES)

Background

Dams or 'impoundments' directly influence reptile populations, especially turtles, by fragmenting habitat and causing nest mortality through flooding via artificial stream regulations (e.g. artificially raising water levels in summer for boat navigation purposes; Bodie 2001). Undamming rivers and restoring ecological flow are both management interventions used to increase river health and change water levels (Bednarek 2001).

Bednarek A.T. (2001) Undamming rivers: a review of the ecological impacts of dam removal. *Environmental Management*, 27, 803–814.

Bodie J.R. (2001) Stream and riparian management for freshwater turtles. *Journal of Environmental Management*, 62, 443–455.

A controlled, before-and-after study in 1997–2013 in two marshes with canals in Camargue, France (1) found that autumn–spring flooding along with autumn–winter grazing increased European pond turtle *Emys orbicularis* abundance. European pond turtle abundance was greater with autumn–spring marsh flooding and moderate-stocking density autumn–winter grazing (192–436 individuals), compared to winter–spring marsh flooding and high-stocking density spring–summer grazing (107–182 individuals) or year-round flooding and low-stocking density grazing (182–227 individuals). In a nearby site with moderate year-round flooding and grazing, European pond turtle abundance was stable over the same time period (29–153 individuals, data taken from graphs). Turtles were live-trapped in April–August 1997–2013 (7,059 total captures of 963 individuals) in two sites (100–250 ha, 1.5 km apart). In 1997–2001, both sites were flooded and grazed year-round at low-moderate stocking density. In one site, in 2002–2006, water levels were modified to create a dry period in summer–autumn, with natural flooding in winter–spring and grazing was changed to high density stocking in spring–summer (see original paper for details). In the same site, in 2007–2013, the flooding period was extended so that autumn–spring were flooded and only summer was dry, and moderate density grazing took place in autumn–winter.

(1) Ficheux S., Olivier A., Fay R., Crivelli A., Besnard A. & Bechet A. (2014) Rapid response of a long-lived species to improved water and grazing management: The case of the European pond turtle (*Emys orbicularis*) in the Camargue, France. *Journal for Nature Conservation*, *22*, 342–348.

8.8. Alter water flow rates

- **One study** evaluated the effects of altering water flow rates on reptile populations. This study was in Australia[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Abundance (1 study):** One before-and-after study in Australia[1] found that releasing a large flow of water into a wetland system had mixed effects on relative abundance of eastern long-necked turtles and the number of turtles caught.
- **Condition (1 study):** One before-and-after study in Australia[1] found that after releasing a large flow of water into a wetland system, body condition of eastern long-necked turtles improved.

BEHAVIOUR (0 STUDIES)

Background

Changes to natural stream flows, for example channelization, may negatively impact reptiles due to faster moving water (Bodie 2001), and changes in summer flow rates may alter the time available for nesting, impact the timing of hatching and affect survival rates (Lenhart *et al.* 2013). Interventions aimed at recovering natural flows, for example by removing the structures causing channelization and increasing flow rates or providing areas that have varied flow rates may address this threat.

Bodie J.R. (2001) Stream and riparian management for freshwater turtles. *Journal of Environmental Management*, 62, 443–455.

Lenhart C.F., Naber J.R. & Nieber J.L. (2013) Impacts of hydrologic change on sandbar nesting availability for riverine turtles in Eastern Minnesota, USA. *Water*, 5, 1243–1261.

A before-and-after study in 2008 in an area of wetland and creeks along a river in south-eastern Australia (1) found that after a large flow of water was released into the system the relative abundance of eastern long-necked turtles *Chelodina longicollis* remained similar, and turtle body condition improved. Following the provision of a large flow of water, a similar number of turtles were caught in a refuge pool (0.02 turtles/trap/hour) compared to before (0.15 turtles/trap/hour). However, authors reported that only six turtles were caught after the water flow, whereas 44 were caught before. Turtles in the refuge pool had higher body condition after the flow than before (reported as condition index). In 2008, water flow into Barmah National Park was increased by opening regulators along a river that feeds into the wetlands and creeks of the park. This released 300 ML of water into the system. One pool was surveyed for turtles in October 2008, just prior to the water release, and again in February 2009, three months after the water release. Turtles were trapped using six fyke nets (50 mm mesh) that were set in shallow water in the early evening and retrieved the following morning. Captured turtles were measured and released.

(1) Howard K., Beesley L., Ward K. & Stokeld D. (2017) Preliminary evidence suggests freshwater turtles respond positively to an environmental water delivery during drought. *Australian Journal of Zoology*, 64, 370–373.

8.9. Maintain dams or water impoundments

- **One study** evaluated the effects of maintaining dams or water impoundments on reptile populations. This study was in the USA[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One replicated, before-and-after study in the USA[1] found that after sediment removal, or dam maintenance along with sediment removal, one water impoundment was still used by Sonoran mud turtles and a second was not used.

Background

Dams or 'impoundments' may provide functional habitat for reptiles. Maintaining dam structures, including removing silt may benefit reptiles that depend on impounded water sources.

See also: *Modify dams or water impoundments to enable wildlife movements.*

A replicated, before-and-after study in 1994–2013 of three intermittent water impoundments in Arizona and New Mexico, USA (1) found that following silt removal and (in one case) dam repair, Sonoran mud turtles *Kinosternon sonoriense* still used one restored pond but were not seen at a second. After silt was removed and a dam leakage repaired at one water impoundment, Sonoran mud turtles were caught in similar numbers to before the dam started leaking (no data are provided). At a second water impoundment, no turtles were caught after sediment was removed, although eight turtles had been caught within 0.1 km of the site previously. At a third water impoundment, the pond re-silted within two months of being cleared and no results for Sonoran mud turtles were reported. Sediment (88–190 m^3) was mechanically removed from three water impoundments in May 2012. The first impoundment (375 m^2 surface area) had dam leakages and drained completely in 2008 and again in 2011 and was repaired in 2009 and 2012. The second impoundment (95 m^2 surface area after de-silting) was completely silted and dry from 1994. The third impoundment (300 m^2 surface area) was filled with silt and wildfire ash from 2010. Surveys for turtles took place in 1994–2013 (no details are provided).

(1) Stone P.A., Congdon J.D. & Smith C.L. (2014) Conservation triage of Sonoran mud turtles (*Kinosternon sonoriense*). *Herpetological Conservation and Biology*, 9, 448–453.

8.10. Modify dams or water impoundments to enable wildlife movements

- **One study** evaluated the effects on reptile populations of modifying dams or water impoundments to enable wildlife movements. This study was in the USA[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One study in the USA[1] found that an eel ladder was used by common watersnakes in five of eight years.

Background
Dams or 'impoundments' may restrict aquatic reptile movements. Modifying dam structures, for example by providing ladders, may enable reptiles to move up and down stream of dam sites.

See also: *Maintain dams or water impoundments*.

A study in 2007–2014 on a river in West Virginia, USA (1) found that an eel ladder was used by common watersnakes *Nerodia sipedon* in five of eight years of monitoring. The ladder was used by common watersnakes (1–5 individuals/year) in five of eight years that the ladder was monitored. A stainless steel fish ladder (11 m long, 13 cm deep and 41 cm wide with a 50° slope containing a suitable substrate for climbing, see original paper for details), designed to facilitate the upstream passage of the snake-like movements of American eels *Anguilla rostrata*, was installed from late spring (May–July) to autumn (October–November) in 2007–2014 (106–188 days/year). Numbers of snakes (and eels) were monitored by live catching or photographs when they reached the upstream end of the ladder.

(1) Welsh S.A. & Loughman Z.J. (2015) Upstream dam passage and use of an eel ladder by the common watersnake (*Nerodia sipedon*). *Herpetological Review*, 46, 176–179.

Other natural system modifications

8.11. Restore or maintain beaches ('beach nourishment')

- **Three studies** evaluated the effects of restoring or maintaining beaches on reptile populations. All three studies were in the USA[1,2,3].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (3 STUDIES)

- **Abundance (1 study):** One replicated, site comparison study in the USA[3] found that gopher tortoise densities were higher and numbers occupying burrows similar on constructed sand dunes compared to natural dunes.
- **Reproductive success (2 studies):** Two controlled, before-and-after studies in the USA[1,2] found that one year after adding sand to beaches, nesting activity decreased more for loggerhead turtles[1], and loggerhead and green turtles[2] compared to on unmodified beaches. Two years after nourishment, both studies found that loggerhead nesting activity had increased[1,2], and in one study nesting had returned to pre-nourishment levels[2].

BEHAVIOUR (2 STUDIES)

- **Use (1 study):** One replicated, site comparison study in the USA[3] found that burrows on a constructed dune were discovered by gopher tortoises after three months.
- **Behaviour change (1 study):** One controlled, before-and-after study in the USA[1] found that one year after adding sand to beaches, loggerhead turtles made more non-nesting crawls than on unmodified beaches, but the difference was smaller two years after nourishment.

Background

Building soft structures ('beach nourishment') such as sand dunes, or filling in eroded beaches with replacement sand may counteract coastal erosion, increase resilience to storms and protect wildlife (Nordstrom *et al.* 2000, Klein *et al.* 2001). Nourished beaches may also provide nesting sites for sea turtles and so the construction process and the type of fill should be carefully considered to as to maintain the natural beach erosion cycle and provide suitable habitat (e.g. Speybroeck *et al*. 2006).

Nordstrom K.F., Lampe R. & Vandemark L.M. (2000) Re-establishing naturally functioning dunes on developed coasts. *Environmental Management,* 25, 37–51.

Klein R.J., Nicholls R.J., Ragoonaden S., Capobianco M., Aston J. & Buckley E.N. (2001) Technological options for adaptation to climate change in coastal zones. *Journal of Coastal Research*, 17, 531–543.

Speybroeck J., Bonte D., Courtens W., Gheskiere T., Grootaert P., Maelfait J.P., Mathys M., Provoost S., Sabbe K., Stienen E.W.M., Van Lancker V., Vincx M. & Segraer S. (2006) Beach nourishment: an ecologically sound coastal defence alternative? A review. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 16, 419–435.

A controlled, before-and-after study in 1992–1996 of a beach in Florida, USA (1) found that raising the height of a beach ridge ('beach nourishment') decreased loggerhead turtle *Caretta caretta* nesting frequency and increased the frequency of non-nesting crawls, although the effect reduced in the second year after implementation. In the first year after nourishment took place, sea turtle nesting frequency declined more, and non-nesting crawl frequency increased more on the nourished beach compared to unmodified beaches (nesting frequency declined by 4–5 nests/km/day more and non-nesting crawls increased by 5–6 crawls/km/day more on nourished beaches). In the second year, the reduction in nesting was again greater and the increase in non-nesting crawls higher on nourished compared to unmodified beaches, but the size of the effects were smaller and only statistically significant compared to one of the two unmodified beaches (nesting frequency declined by 1–2 nests/km/day more and non-nesting crawls increased by 1 crawl/km/day more on nourished beaches). In March and April 1995, a 1.6 km stretch of beach was nourished with additional sand, increasing the height of the beach ridge from an average of 32 m to 81 m. Sea turtle nesting activity was recorded daily from May to August from 1992 to 1996 at the nourished and two natural beaches three seasons prior to and two seasons immediately following beach nourishment.

A controlled, before-and-after study in 2000–2003 on a sandy beach in Florida, USA (2) found that nesting success for loggerhead *Caretta caretta* and green turtles *Chelonia mydas* declined in the year following beach nourishment, but returned to pre-nourishment levels for loggerheads in the second year following nourishment. Nesting success declined following nourishment for loggerheads (1 year post-nourishment: 30% success; 1 year pre-nourishment: 60% success) and green turtles (1 year post-nourishment: 29%; 2 years pre-nourishment: 64%). Declines in nourished areas were larger than those seen in non-nourished areas over the same period (loggerheads: 63% vs 50%; green turtles: 55% vs 51%). In the second year following nourishment, loggerhead nesting success returned to around pre-nourishment levels (54% success). In 2002, a 5 km stretch of a 40 km beach was artificially nourished with 917,000 m^3 of sand just prior to the start of the nesting season. In May–August 2000–2003, nesting activity was monitored by counting turtle emergence tracks on the beach, and nesting success was defined as the percentage of emergences that resulted in nests.

A replicated, site comparison study in 2012–2016 on four sand dunes in Florida, USA (3) found that gopher tortoises *Gopherus polyphemus* colonized man-made sand dunes within three months of their construction, and densities were higher but occupancy of burrows similar compared to natural dunes. Overall, gopher tortoise density was higher on constructed sand dunes (2012 dune: 21 tortoises/ha; 2014 dune: 2–3) than natural dunes (0–8 tortoises/ha, see statistical model results in paper for more details). The first burrow on the dune built in 2014 was discovered three months after construction. Burrow occupancy rates were similar between dunes (2012 dune: 0.6 tortoises/burrow; 2014 dune: 0.4; natural dunes: 0.3). Gopher tortoise use of two natural and two constructed sand dunes (built in 2012 and 2014) was evaluated by surveying a 3 km long stretch of beach for tortoise burrows in May–August 2014 and 2015 (twice/year) and January 2015 and 2016 (once/year). Gopher tortoise burrow occupancy was assessed using cameras in 20 randomly selected burrows each January (one–two surveys/burrow). Resident tortoises were relocated during dune construction.

(1) Rumbold D.G., Davis P.W. & Perretta C. (2001) Estimating the effect of beach nourishment on *Caretta caretta* (loggerhead sea turtle) nesting. *Restoration Ecology*, 9, 304–310.

(2) Brock K.A., Reece J.S. & Ehrhart L.M. (2009) The effects of artificial beach nourishment on marine turtles: differences between loggerhead and green turtles. *Restoration Ecology*, 17, 297–307.

(3) Martin S.A., Rautsaw R.M., Bolt R., Parkinson C.L. & Seigel R.A. (2017) Adapting coastal management to climate change: mitigating our shrinking shorelines. *Journal of Wildlife Management*, 81, 982–989.

8.12. Armour shorelines to prevent erosion

- We found no studies that evaluated the effects of armouring shorelines to prevent erosion on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Shorelines are armoured to prevent shoreline retreat, a natural process where waves erode coastlines. These physical structures range from relatively temporary systems such as sandbags to permanent structures such as rock walls, seawalls, living seawalls and offshore breakwaters. Armouring shorelines may benefit reptiles if it leads to the protection of habitats used by reptiles (e.g. nesting beaches) that would otherwise be lost to erosion.

9. Threat: Invasive alien and other problematic species

Invasive and other problematic species of animals, plants and diseases are known to have caused species extinction worldwide (Bellard *et al*. 2016), and this threat is listed as a key cause of decline or extinction for a number of reptile species that are extinct [Cape Verde giant skink *Chioninia coctei* (Vasconcelos 2013), Navessa rhinoceros iguana *Cyclura onchiopsis* (Powell 2011), Tonga ground skink *Tachygyia microlepis* (Allison *et al.* 2012)] or extinct in the wild [Christmas Island Chained Gecko *Lepidodactylus listeri* (Cogger *et al.* 2017), Christmas Island blue-tailed shinning-skink *Cryptoblepharus egeriae* (Woinarski *et al.* 2017)]. Introduced animals including mongooses *Herpestes auropunctatus*, rats *Rattus* spp., pigs *Sus scrofa,* and goats *Capra hircus*, as well as dogs *Canis familiaris* (Weston & Stankowich 2013) and cats *Felis catus* (Medina *et al*. 2011) have all been implicated in species declines and extinctions. Invasive species may prey on reptiles and their eggs, compete for resources, and alter habitats, all of which can lead to population declines as has occurred with the tuatara *Sphenodon spp.*, pelagic gecko *Nactus pelagicus*, and Galápagos tortoises *Chelonoidis nigra*-complex (Gibbons *et al*. 2000).

This chapter describes the evidence from interventions designed to reduce the threat from invasive and other problematic species.

For studies that discuss the effects of relocating nests, including those that relocate nests to protect them from predation, see *Species management – Relocate nests/eggs to a hatchery*; *Relocate nests/eggs to a nearby natural setting (not including hatcheries)* and *Relocate nests/eggs for artificial incubation*.

Allison A., Hamilton A. & Tallowin O. (2012) *Tachygyia microlepis*. *The IUCN Red List of Threatened Species* 2012: e.T21286A2775072. Accessed 10 November 2021.

Bellard C., Cassey P. & Blackburn T.M. (2016) Alien species as a driver of recent extinctions. *Biology Letters*, 12, 20150623.

Cogger, H., Mitchell, N.M & Woinarski, J. 2017. *Lepidodactylus listeri*. *The IUCN Red List of Threatened Species* 2017: e.T11559A83321765. Accessed 10 November 2021.

Gibbons J.W., Scott D.E., Ryan T.J., Buhlmann K.A., Tuberville T.D., Metts B.S., Greene J.L., Mills T., Leiden Y., Poppy S. & Winne C.T. (2000) The global decline of reptiles, déjà vu amphibians. *BioScience*, 50, 653–666.

Medina F.M., Bonnaud E., Vidal E., Tershy B.R., Zavaleta E.S., Josh Donlan C., Keitt B.S., Corre M., Horwath S.V. & Nogales M. (2011) A global review of the impacts of invasive cats on island endangered vertebrates. *Global Change Biology*, 17, 3503–3510.

Powell R. (2011) *Cyclura onchiopsis*. *The IUCN Red List of Threatened Species* 2011: e.T173001A6955940. Accessed 10 November 2021.

Vasconcelos R. (2013) *Chioninia coctei*. *The IUCN Red List of Threatened Species* 2013: e.T13152363A13152374. Accessed 10 November 2021.

Weston M.A. & Stankowich T. (2013) Dogs as agents of disturbance. Pages 94–113 in: M.E. Gompper, (eds.) *Free-Ranging Dogs and Wildlife Conservation.* Oxford University Press.

Woinarski, J., Cogger, H., Mitchell, N.M & Emery, J. 2017. *Cryptoblepharus egeriae*. *The IUCN Red List of Threatened Species* 2017: e.T102327291A102327566. Accessed 10 November 2021.

Reduce predation by other species

9.1. Remove or control predators using lethal controls

Background

Predators can drive declines or local extinctions of vulnerable reptile species. Non-native predators may be a particular problem for native reptiles that lack sufficient predator avoidance behaviours. Native predators can also threaten populations of reptiles that persist in low numbers. Removing or controlling predators, especially native predators, for the benefit of their wild prey species can be a controversial management strategy. Nonetheless, there is potential for such management to lead to increases in the abundance, survival or reproductive success for species of conservation concern.

Due to the number of studies found, this action has been split by species group, though no studies were found for amphisbaenians.

See also: *Remove or control predators using fencing and/or aerial nets* and *Remove or control predators by relocating them*.

Sea turtles

- **Four studies** evaluated the effects of removing or controlling predators using lethal controls on sea turtle populations. All four studies were in the USA[1-4].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (4 STUDIES)

- **Reproductive success (4 studies):** Two before-and-after studies (including one controlled study) in the USA[3,4] found that on islands where raccoons and feral pigs[3] or only feral pigs[4] were eradicated, fewer loggerhead[3] and loggerhead and green turtle nests[4] were predated than before predator control began. One replicated, randomized, controlled study in the USA[1] found that controlling raccoons on short sections of a beach resulted in similar predation of loggerhead turtle nests compared to in sections of the beach with no control. One before-and-after study in the USA[2] found that disruptions to a programme controlling raccoons and armadillos resulted in more predation of loggerhead, leatherback and green turtle nests.

BEHAVIOUR (0 STUDIES)

A replicated, randomized, controlled study in 1993–1994 on a long sandy beach in Florida, USA (1) found that raccoon *Procyon lotor* control on sections of beach did not reduce predation of loggerhead turtle *Caretta caretta* nests compared to sections with no control. Predation of turtle nests was similar in sections of the beach with raccoon control (1993: 489 of 1,359 nests, 36% predation; 1994: 460 of 1,686, 27%) and those with no control (1993: 72 of 231 nests, 31%; 1994: 92 of 379, 24%). In 1993–1994, a long stretch of barrier beach (37 km) was broken down into four experimental blocks, and around half of each block (4 km) was selected for raccoon removal. Raccoons were captured with live traps baited with sardines, anesthetised and lethally injected (215 individuals; estimated as 50% of raccoon population). Nests in the remainder of the block were either received no treatment or were part of further tests of the effect of nest screening and taste aversion on predation. Nests were monitored 2–4 times/month in 1993 and 2–3 times/week in 1994.

A before-and-after study in 2002–2004 on a sandy beach in Florida, USA (2) found that disruptions to the control of raccoons *Procyon lotor* and invasive armadillos *Dasypus novemcinctus* resulted in reduced survival of loggerhead *Caretta caretta*, leatherback *Dermochelys coriacea* and green turtle *Chelonia mydas* nests due to predation. In 2002–2004, months when predator control was consistent across all years (May), nest survival was similar (>80% after 80 days). However, disruptions to control in June 2004 resulted in lower survival for nests laid in June–July 2004 (60–70% after 60–80 days) compared to June–July 2002–2003 (>80% after 60–80 days). In 2002–2004, raccoons were live trapped and killed, and both raccoons and armadillos were shot (0.22 calibre rifle with a noise suppressor and night-vision equipment). In 2004, predator control ceased for 2 weeks in June, re-started in July, and then ended completely in August. The beach was monitored daily starting in March 2002–2004, and all leatherback and green turtle nests were marked, but only every eighth loggerhead nest marked and monitored. Marked nests were monitored daily for predation and excavated after hatchling emergence to assess hatching success.

A controlled, before-and-after study in 2006–2008 on two barrier islands in Florida, USA (3) found that eradication of racoons *Procyon lotor* and feral pigs *Sus scrofa* resulted in lower predation rates of loggerhead turtle *Caretta caretta* nests compared to sites with no predator control. Predation of sea turtle nests was lower following predator removal (0–16% of 2–143 nests) compared to before predator removal began (60–84% of 20–76 nests). Two study islands were established with no predator control taking place in 2006. Control of raccoons and feral swine began in 2007 on one island and in 2008 on the other. Raccoons and pigs were captured in baited traps, and free-roaming pigs were also shot with a noise suppressed rifle. Sea turtle nests were monitored by patrolling beaches in the morning during the turtle nesting season noting new nests and evidence of predation.

A before-and-after study in 2007–2010 on a sandy beach on an island off the coast of Florida, USA (4) found that eradicating feral pigs *Sus scrofa* ended pig predation of unhatched loggerhead *Caretta caretta* and green turtle *Chelonia mydas* nests. In the nesting season after feral pigs were eradicated, no marine turtle nests were lost to pig predation, compared to 50 of 50 nests predated by pigs in the nesting season prior to their eradication, including 36 nests covered with nest cages. In 2005–2010, turtle nests were monitored daily throughout the nesting season on a 13.4 km long stretch of beach on an island (526 ha). In 2007, nest predation by pigs was observed for the first time (pigs were present on the island from 2001). In May–June 2008, a total of 39 feral pigs were eradicated from the island by trapping and shooting over bait. Following eradication, no spoor or other signs of feral pigs were found. The authors reported that pigs reinvaded the island in 2014. Thirty-six of the 50 nests monitored had been covered by partially buried cages (91 cm long x 91 cm wide x 76 cm tall, 5 x 10 cm wire mesh) to protect them from raccoon predation. After hatching, all nests were excavated to record hatching success and predation levels. See 'Use nest cages' for more details on their effectiveness.

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(2) Engeman R.M., Martin R.E., Smith H.T., Woolard J., Crady C.K., Constantin B., Stahl M. & Groninger, N.P. (2006) Impact on predation of sea turtle nests when predator control was removed midway through the nesting season. *Wildlife Research*, 33, 187–192.

(3) Engeman R.M., Duffiney A., Braem S., Olsen C., Constantin B., Small P., Dunlap J. & Griffin J.C. (2010) Dramatic and immediate improvements in insular nesting success for threatened sea turtles and shorebirds following predator management. *Journal of Experimental Marine Biology and Ecology*, 395, 147–152.

(4) Engeman R.M., Addison D. & Griffin J.C. (2016) Defending against disparate marine turtle nest predators: nesting success benefits from eradicating invasive feral swine and caging nests from raccoons. *Oryx*, 50, 289–295.

Tortoises, terrapins, side-necked & softshell turtles

- **Seven studies** evaluated the effects of removing or controlling predators using lethal controls on tortoise, terrapin, side-necked and softshell turtle populations. Four studies were in the USA[3-5,7] and three were in Australia[1,2,6].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (7 STUDIES)

- **Reproductive success (7 studies):** Six of seven studies (including four replicated, controlled studies) in Australia[1,2,6] and the USA[3-5,7] found that in areas with mammal[1-4,7] or fire ant[5] control, and in two cases with fencing[4,5], fewer tortoise[4,5], turtle[1,2,7] and terrapin[3] nests were predated compared to areas with no control, or before control began. Two studies[3,7] also found that predation increased again a year after control or in the second year of control. The other study[6] found that following short-term fox control, a similar number of artificial eastern long-necked turtle nests were predated by foxes compared to before control began.
- **Survival (3 studies):** Two of three replicated, controlled studies (including one before-and-after study and one randomized study) in Australia[2] and the USA[4,5] found that in a fenced area with mammal[4] or fire ant control[5], more gopher tortoise hatchlings survived for one year[4] or at least 150 days[5] compared to fenced areas with no control. The other study[2] found mixed effects of fox control on survival of Murray short-necked turtles and broad-shelled turtles depending on turtle species, age and sex.

BEHAVIOUR (1 STUDY)

- **Behaviour change (1 study):** One replicated, controlled, before-and-after study in Australia[1] found that in areas with fox control, freshwater turtles nested further from the water and nests were more spread out compared to areas with no control, or before control began.

A replicated, controlled, before-and-after study in 1996–2000 in an area with four lagoons in south-eastern Australia (1, same experimental set-up as 2) found that removing foxes *Vulpes vulpes* resulted in lower predation rates of Murray short-necked river turtle *Emydura macquarii* nests and changes in nesting behaviour compared to when foxes were present. Nest predation was lower after fox removal (<50% nests predated) compared to areas with no fox removal (>85%) and before fox removal started (85–93% of 12–29 nests predated). Following fox removal, turtles nested further from the water (25–26 m) compared to before removal and no-removal sites (14–19 m), and nests were more spread out (removal: 12–16 m between nests; no removal: 8–11 m). In May 1997 to January 1999, fox control was carried out at two lagoons by burying poison baits (35 g FOXOFF baits) along fence lines, hill ridges and access roads (150–200 m apart, 48 baits/site; laid every 1–2 months) and shooting foxes. A further two lagoons had no fox removal. Searches for turtle nests were conducted in November 1996–1998.

A replicated, controlled, before-and-after study in 1996–2000 in an area with four lagoons in south-eastern Australia (2, same experimental set-up as 1) found that removing foxes *Vulpes vulpes* resulted in higher nesting success of Murray short-necked river turtles *Emydura macquarii* and broad-shelled turtles *Chelodina expansa* and higher survival of female short-necks compared to areas with no removal. Nest predation was lower after fox removal (short-necked turtles: <50% nests predated; broad-shelled: 18–38% predated) compared to areas with no fox removal (short-necked: >85% predated; broad-shelled: 57 and 50% of 7 and 10 nests) and before fox removal started (short-necked: 85–93% of 12–29 nests predated; 55–70% of 8–11 nests) (results were not statistically tested). Survival of female short-necks was higher following fox removal (97–98% survival) compared to areas with no removal (93–95%) and before fox removal (94 and 95%), though no effects of fox removal were found for short-neck males (95–99% survival), juveniles (69%), or any group of broad-shelled turtles (84–92%). In May 1997 to January 1999, fox control was carried out at two lagoons by burying poison baits (35 g FOXOFF baits; 150–200 m apart; 48 baits/site, laid every 1–2 months) and shooting foxes. A further two lagoons had no fox removal. In 1996–1998, searches for turtle nests were conducted in late autumn and trapping was conducted every 14–18 days in September–March.

A before-and-after study in 1997–2000, and 2005–2006 on an intracoastal island in Florida, USA (3) found that removing raccoons *Procyon lotor* resulted in reduced predation of Carolina diamondback terrapin *Malaclemys terrapin centrata* nests compared to when no removal was carried out. Nest predation was lower in the year with racoon removal (7 of 93, 8% of nests predated) than in years prior to removal (1997: 61 of 114, 54%; 2000: 57 of 112, 51%). Predation increased again in the year after removal (39 of 45, 87%). Racoons were trapped daily from February to April 2005, and then intermittently until September 2005 using live traps (23 raccoons removed). Raccoons were anesthetised and euthanised by lethal injection. Beaches were searched daily from April–October for signs of nesting turtles, and these nests were then monitored daily for signs of predation and emerging young.

A replicated, controlled study in 2002–2005 in a pine forest in Georgia, USA (4) found that removing predators from fenced exclosures resulted in higher survival of gopher tortoise *Gopherus polyphemus* nests and hatchlings compared to areas with no fencing or predator removal. The effects of predator removal (lethal controls and relocations) and fencing cannot be separated. Survival was higher inside fenced areas with predator removal compared to outside for both nests (fenced: 52 of 78, 66% survived; unfenced: 26 of 73, 35%) and hatchlings (fenced: 74% survived for 1 year; unfenced: 38%). In 2002–2003, four plots (40 ha) were randomly selected and enclosed in 1.1m high mesh fence with electrical wires at the top and bottom. A further four plots were left unfenced. In 2002–2003, all mammalian predators within the exclosures were live-trapped and relocated, and in 2003–2005, further trapping of predators was conducted. Predators that re-entered exclosures were euthanized. In May–June 2003–2005, all tortoise burrows were searched for nests, and all active nests were monitored 1–2 times/week up to 110 days. In 2004, forty hatchlings from 13 different nests were fitted with radio transmitters and monitored for up to a year.

A replicated, randomized, controlled study in 2014–2015 in mixed forest and agricultural land in Georgia, USA (5) found that when fire ants *Solenopsis invicta* were controlled with insecticide, gopher tortoise *Gopherus polyphemus* nests were not predated and hatchling survival rates increased. None of 16 gopher tortoise nests were predated by fire ants in insecticide-treated enclosures, compared to eight of 16 nests in untreated enclosures. Hatchling survival was higher in insecticide-treated enclosures (16 of 16 individuals survived at least 150 days) compared to untreated enclosures (5 of 16 individuals survived; five were killed by fire ants and six by raccoons *Procyon lotor*). Fire ant abundance were reduced in insecticide-treated enclosures (0.3–10.0 fire ants) compared to unmanaged enclosures (122–537 fire ants). In May–June 2014, wild-laid gopher tortoise nests were relocated to eight fenced 0.2 ha enclosures (four nests/enclosure, two eggs/nest, 64 total eggs). All nests were covered with cloth cages (30 x 30 x 12 cm). Four of eight enclosures were treated with Amdro® insecticide (1.7 kg/ha) to reduce fire ant numbers. Fire ants were monitored using baited traps. Nests were monitored weekly until two weeks before expected emergence, daily thereafter and excavated after 120 days. Hatchlings were radio tracked (16 individuals each from insecticide-treated and untreated enclosures) from August 2014 to March 2015.

A before-and-after study in 2014–2015 around four lakes in northwest Victoria, Australia (6) found that short-term fox *Vulpes vulpes* control did not reduce predation on artificial eastern long-necked turtle *Chelodina longicollis* nests. The number of artificial nests predated by foxes was similar following short term fox control (78 of 95, 82% of nests) compared to before control (59 of 70, 84%). In November 2014, twenty-one days of fox control was implemented by burying baits (1080/sodium monofluoroacetate) across 175 bait stations (25,000 ha site). Artificial nests were randomly placed around 14 sites along the shores of four lakes in sandy soil, 5–30 m from the lake's edge (70 nests pre-control; 95 nests post-control). Each nest consisted of a hand-dug boot-shaped chamber 10–15 cm deep with five quail eggs sprayed with water from captive turtle ponds and covered with sand and surface litter. Nests were inspected four times (up to 35–41 days after construction) for signs of predation by foxes.

A before-and-after study in 2013–2014 in mixed sandy grassland, woodland and marsh habitats along the Illinois-Wisconsin state borders, USA (7) found that removing raccoons *Procyon lotor* led to less Blanding's turtle *Emydoidea blandingii* nest predation in the first year, but not in the second year, of predator management. Results were not statistically tested. In 2013, one of seven (14%)

Blanding's turtle nests were partially predated and no nests were completely predated. In 2014, nine of 15 (60%) of turtle nests were predated (one partially, eight completely predated). The authors reported that before predator management, 12 of 13 (92%) of turtle nests and 88% of monitored artificial nests were predated (see original paper for details). In April-May 2013 and 2014, a total of 78 raccoons (an estimated 83–89% of the total population) were trapped and euthanized in a designated nature reserve and adjacent areas (338–389 ha). In 2013–2014, twenty-two gravid female turtles were captured and monitored closely until egg laying using radio-telemetry. Nests were marked and monitored daily for evidence of excavation or predation. Turtle nests were not protected.

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(2) Spencer R.J. & Thompson M.B. (2005) Experimental analysis of the impact of foxes on freshwater turtle populations. *Conservation Biology*, 19, 845–854.
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(4) Smith L.L., Steen D.A., Conner L.M. & Rutledge J.C. (2013) Effects of predator exclusion on nest and hatching survival in the gopher tortoise. *The Journal of Wildlife Management*, 77, 352–358.
(5) Dziadzio M.C., Chandler R.B., Smith L.L. & Castleberry S.B. (2016) Impacts of red imported fire ants (*Solenopsis invicta*) on nestling and hatchling gopher tortoises (*Gopherus polyphemus*) in southwest Georgia, USA. *Herpetological Conservation and Biology*, 11, 527–538.
(6) Robley A., Howard K., Lindeman M., Cameron R., Jardine A. & Hiscock D. (2016) The effectiveness of short-term fox control in protecting a seasonally vulnerable species, the eastern long-necked turtle. *Ecological Management & Restoration*, 17, 63–69.
(7) Urbanek R.E., Glowacki G.A. & Nielsen C.K. (2016) Effect of raccoon (*Procyon lotor*) reduction on Blanding's turtles (*Emydoidea blandingii*) nest success. *Journal of North American Herpetology*, 39–44.

Snakes & lizards

- **Twelve studies** evaluated the effects of removing or controlling predators using lethal controls on snake and lizard populations. Four studies were in New Zealand[1,2,6,10], two were in each of Australia[3,5] and the Galápagos[7,12], and one was in each of Indonesia[4], Antigua[8], Mexico[9] and the Bahamas[11].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (12 STUDIES)

- **Abundance (8 studies):** Four of six before-and-after studies (including one replicated, controlled study) in New Zealand[1,2,6], Antigua[8], Mexico[9] and the Bahamas[11] found that on islands where both Pacific rats and European rabbits[2], Pacific rats[6], black rats[8] and cats[9] were eradicated, the abundance of lizards[2,6,9] and Antiguan racer snakes[8] increased. One study[11] found that on an island where black rats were eradicated the number of San Salvador rock iguanas remained similar compared to before eradication. The other study[1] found that eradicating mice had mixed effects on the abundance of lizards. One study[2] also found that lizard abundance on an island with eradication was initially lower than on a predator free island, but after two years was similar or higher. One controlled, before-and-after study in Australia[3] found that across areas with fox and cat control or only fox control, gecko and skink numbers were similar to an area with no control, but dragon lizard numbers were lower. One replicated, site comparison study in Australia[5] found that in areas with fox control sand goanna abundance was higher and there was mixed effects on small lizard abundance compared to in areas with no control.

- **Reproductive success (1 study):** One before-and-after study in the Galápagos[7] found that on an island where cats were eradicated the number of offspring of reintroduced Galápagos land iguanas was higher than before cat control began.
- **Survival: (2 studies):** One study in New Zealand[10] found that survival of captive-bred Otago skinks released into an enclosure after mouse eradication was higher compared to when skinks were released in the presence of mice. One study in Indonesia[4] reported no mortality of monitor lizards following use of poison baits to control black rats.
- **Condition (2 studies):** One of two studies in Indonesia[4] and the Galápagos[12] found that on an island where black rats were controlled, rodenticide was detected in the livers of lava lizards for up to 850 days after its use began[12]. The other study[4] reported no illness in monitor lizards following use of poison baits to control black rats.

BEHAVIOUR (0 STUDIES)

A before-and-after study in 1986–1993 on Mana Island, New Zealand (1) found that following eradication of an invasive house mouse *Mus musculus*, the abundance of two of four lizard species increased, and two remained stable. Before-and-after comparisons were not statistically tested. In the four years following mouse eradication, the number of McGregor's skinks *Cyclodina macgregori* increased from one capture/100 trap nights in the year after eradication to 10 captures/100 trap nights three years after eradication. Numbers prior to eradication had been six captures (four years before), 8 captures (2–3 years before) and one capture/100 trap nights (one years before). Common gecko *Hoplodactylus maculatus* captures increased following eradication (after: 35–70 captures/100 trap nights; before: 5–15). Numbers captured remained similar for common skinks *Leiolopisma nigriplantare polychroma* (after: 6–10 captures/100 trap nights; before: 9–21) and copper skinks *Cyclodina aenea* (after: 2–4 captures/100 trap nights; before: 1–9). In 1989–1990, mouse eradication was carried out by distributing poison baits (Storm, Talon 20P, Talon 50W) via two aerial drops and ground baiting (over 5,000 stations in 25 m). In 1986–1987, cattle were also removed from the island. In 1985–1993, lizards were trapped annually (3–8 sessions/year; 2–4 days trapping/session) using pitfall traps (582–4,066 trap nights/session) that were deployed across 27 trapping stations around the island.

A before-and-after, site comparison study in 1986–1993 on two islands near North Island, New Zealand (2) found that removal of Pacific rats *Rattus exulans* and European rabbits *Oryctolagus cuniculus* resulted in an increase in the abundance of lizards, and when compared to a predator free island, abundance was initially lower but after two years was similar or higher on the removal island. The effects of predator and herbivore control cannot be separated. In forest sites, lizard numbers remained stable for five years following eradication (1986–1991: 2 lizards/100 trap days) before increasing suddenly (1992–1993: 16 lizards/100 trap days), and in coastal sites there was a gradual increase from the year of eradication (3 lizards/100 trap nights) to six years after eradication (70 lizards/100 trap nights). In 1986–1987, lizard abundance was lower on the removal island compared to a predator free island (coastal areas only: removal: 2–5 lizards/100 trap days; predator free: 16–49), but in 1988–1992, abundance was similar in two (1988: 14–15; 1992: 60–69) and higher in two years (1990–1991: removal: 61–67; predator free: 20–36; no data collected in 1989 on predator free island). Rats and rabbits were eradicated in 1986–1987 from one island (rodenticide and shooting) and a nearby island was historically free of invasive mammals. Lizards were counted using pitfall traps along four transects (20 traps/transect, two each in coastal and forested areas) on the removal island (March and November 1986–1993), and on the predator free island (November 1986–1992).

A controlled, before-and-after study in 1990–1994 in mixed heath and dune habitat in Western Australia, Australia (3) found that where cats *Felis catus* and foxes *Vulpes vulpes*, or just foxes were controlled, captures of reptiles did not increase. The number of geckos and skinks were similar in areas with cat and fox control, fox control only or no control (geckos: 1–2 individuals/trap grid; skinks: 2 individuals/trap grid). Dragon lizard numbers were lower in areas with greater predator control (cat and fox control: 2 individuals/trap grid; fox control only: 5 individuals/trap grid; no predator control: 7 individuals/trap grid). In areas with predator control, there was no clear change in reptile numbers from before control began (0–24 individuals/group/year) compared to the three years after control began (0–12 individuals/group/year). In 1991, a mainland peninsula was divided into three areas: one area (12 km^2) where cats and foxes were controlled (using electrified fencing, poison baiting, or secondary poisoning by poisoning European rabbits *Oryctolagus cuniculus*, trapping or shooting); one area (120–200 km^2) where foxes were controlled by baiting but cats were not targeted; and one area where no control occurred. Reptiles were monitored with six pitfall-trap and drift fence grids in each area (18 in total). Each grid had eight pitfall traps, 30–50 m apart. Sampling was conducted over three consecutive days in March–April and June–July in 1990–1994 in predator control areas and 1992–1994 in the area without predator control. Reptiles captured included dragon lizards, skinks, geckos, snakes, and a species of monitor lizard and blind snake but only species that could be toe clipped (dragon lizards, skinks and geckos) were included in analysis.

A study in 2003 on an island offshore of East-Kalimantan, Indonesia (4) found that carrying out lethal control of black rats *Rattus rattus* using poison baits did not have detrimental effects on monitor lizards *Varanus salvator*. No illness or mortality was recorded in any monitor lizards. The last living rat was observed 5–6 months following the deployment of poison baits. In February 2003, a 25 x 25 m grid was established across the whole island and a bait station placed in each grid square. An additional 23 bait stations were established around the perimeter of the island. Blocks of rodenticide (Klerat®) were deployed at each bait station from the 7th April 2003 and replaced as needed. All non-target species were monitored throughout the baiting period.

A replicated, site comparison study in 1999–2000 in two sites of semi-arid shrubland, grasses and sparse woody plants in New South Wales, Australia (5) found that an area with long term poison baiting for foxes *Canis vulpes* had more sand goannas *Varanus gouldii* compared to an un-baited area, but effects on small lizards were mixed. More goannas were found in the area with fox baiting (52 individuals) compared to the un-baited area (9 individuals). Overall, small lizard abundance was similar between the baited area (0.4 lizards/trap) and un-baited area (0.5 lizards/trap), but in one of three habitat types lizards were less abundant in the baited (0.2/trap) compared to un-baited area (0.4/trap). Skinks were more abundant in the baited areas in one of three habitat types (baited: 0.6/trap; un-baited: 0.2/trap) and geckos were less abundant in baited areas in one of three habitat types (baited: 0.1/trap; un-baited: 0.5/trap), but in all other comparisons abundances were similar. At one site, poison baiting (1080-bait) started in 1995 along roads, and from 1997 three aerial baitings/year were also carried out. An additional site (75 km away) received no baits. In 2000, two areas in each site (baited area: >20 km apart; un-baited area: 15 km apart) were surveyed for sand goannas from a vehicle (580–590 km/site). In November–December 1999 (11 days), at each area in both sites, small reptiles were trapped across three habitat types (grassland, mallee/woodland and spinifex), with three traps lines/trapping location (15 m drift fence, with 5 pit-fall traps).

A before-and-after study in 1992–1996 on a Pacific island off the east coast of North Island, New Zealand (6) found that following Pacific rat *Rattus exulans* eradication, reptile abundances increased. After Pacific rats were eradicated, abundances of skinks, geckos (including the Duvaucel's gecko *Hoplodactylus duvaucelii*) increased (no data are provided). Monitoring after the poison-bait was deployed revealed no signs of rats on the island. Rats were eradicated using aerial-deployed rodenticide bait on Lady Alice Island (145 ha) in 1994 (8 kg brodifacoum-impregnated bait/ha). Reptile monitoring started in 1992, two years before rat eradication and continued for at least two years afterwards. Skinks and geckos were surveyed using pitfall traps.

A before-and-after study in 1999–2003 on a tropical island in the Galápagos, Ecuador (7) found that during an ongoing iguana reintroduction, more offspring of Galapagos land iguanas *Conolophus subcristatus* were captured following a successful cat *Felis catus* eradication program. Results were not statistically tested. The number of offspring of reintroduced iguanas captured was higher after most cats were eradicated (1–14 adults and 6–14 sub-adults and juveniles/year) than before eradication began (0–1 adults and 4–6 sub-adults and juveniles/year). The number of reintroduced iguanas that were recaptured varied throughout the study (after most cats eradicated: 21–32 individuals/year; before eradication: 17–30 individuals/year). Reintroduction efforts were ongoing through the study, with six releases totalling 183 individuals during 1991–2003. In 2001–2003, cat eradication was carried out with poison baits (1080 poison), trapping and shooting, and cats were considered eradicated by 2003. Iguanas were surveyed (6 days in June–July) before (1999–2000) and after (2002–2003) the majority of cat eradication had been completed.

A before-and-after study in 1995–2004 in coastal forest on Great Bird Island, Antigua (8) found that eradicating black rats *Rattus rattus* increased the abundance of Antiguan racer snakes *Alsophis antiguae*. No statistical tests were carried out. The snake population doubled in 2 years after rat eradication compared to before eradication (pre-eradication population estimate: 51 snakes; 2 years post-eradication estimate: 115 snakes) and, although there were between year fluctuations, the snake population remained greater than pre-eradication (population estimates 2–9 years post-eradication: 78–161 snakes). The author reported rat eradication took place on Great Bird Island (10 ha) over three weeks in late 1995. The island was checked monthly for signs of rats after the eradication program ended. In total, 730 bait stations with the rodenticide brodifacoum dispensed in wax blocks were placed in a 10 x 10 m grid across the whole island. Rats were also eradicated from two neighbouring islands. Snakes were surveyed over six weeks in 1995 before rat eradication and annually after rat eradication in 1997–2004. Snake population estimates were calculated using mark-recapture of individual snakes. Rat eradication programme details were sourced from an associated article (Daltry 2006).

Daltry J. (2006) Control of the black rat *Rattus rattus* for the conservation of the Antiguan racer *Alsophis antiguae* on Great Bird Island, Antigua. *Conservation Evidence*, 3, 28–29.

A before-and-after study in 1995–1998 on a tropical island, western Mexico (9) found that following eradication of cats *Felis catus*, the abundance of black iguana *Ctenosaura pectinata* and Clark's spiny lizard *Sceloporus clarkii* increased. The authors reported that after the start of the cat eradication programme, black iguana abundance doubled or quadrupled and that the Clark's spiny lizard was more frequently observed. In 1995–1998, cats (113 individuals/km^2) were eradicated from Isla Isabel (194 ha), by trapping, poisoning (with 1080 sodium monofluoroacetate) and shooting. An attempted eradication of black rats *Rattus rattus* at the same time, using brodifacoum poisoning failed. No details on reptile monitoring were provided.

A study in 2009–2012 in an area of mixed shrub and grassland in Otago, New Zealand (10) found that survival of captive-bred Otago skinks *Oligosoma otagense* released into an enclosure was higher for those released when house mice *Mus musculus* had been eradicated compared to when skinks were released in the presence of mice. Authors reported that post-release survival was higher for skinks released with no mice present (44%) compared to survival of skinks released just prior to reinvasion by mice (15%). Survival of established skinks (2 years after their release) after the mouse reinvasion was higher (91%) than for newly released skinks in the presence of mice (17%). In 2009, a 0.3 ha area was enclosed within a mammal resistant fence (1.9 m high) and over a six-month period, all mammals inside the enclosure were eradicated using a range of baited traps. After eradication, 12 captive-bred adult skinks were released in the enclosure following eight weeks in quarantine. In 2011, an additional 16 skinks were quarantined and released. House mice reinvaded during 2012 and were again eradicated using live capture traps and poison bait stations. In 2009–2012, starting 7–10 days after release, skinks were monitored every 15 days by a walking survey of the enclosure.

A before-and-after study in 1994–2013 on an offshore cay in San Salvador, Bahamas (11) found that using rodenticides to control invasive black rats *Rattus rattus* did not increase the abundance of San Salvador rock iguana *Cyclura rileyi rileyi.* Results were not statistically tested. The authors reported that following the eradication of black rats, abundance of San Salvador rock iguanas did not increase (population estimate after rat eradication: 28–159 iguana; population estimate before 36–144 iguanas). Black rats were controlled using rodenticide (brodifacoum) administered in wax blocks in covered bait stations in 1999 and

2000 (see original paper for detailed methods) on an island (25 acres). In summer 1999, the eradication attempt failed due to bait station design. Eradication was considered successful in summer 2000. San Salvador rock iguana were surveyed across the whole island in 1994, 1998–2007 and 2012–2013 (3 years before eradication and 9 years after eradication) using visual encounter surveys.

A study in 2012–2014 on an island in the Galápagos, Ecuador (12) found that controlling black rats *Rattus rattus* using anticoagulant rodenticides lead to widespread secondary exposure to rodenticides in endemic lava lizards *Microlophus duncanensis*, although no population level impacts were observed. Rodenticide was detected in livers of 270 lizards (brodifacoum concentrations: 10 ppb–2000 ppb in individual livers) and was still being detected up to 850 days after the baiting took place. The authors noted that the secondary exposure of lizards to rodenticides was implicated in the exposure and mortality of 22 Galapagos hawks *Buteo galapagoensis*. Black rat eradication commenced on Pinzon Island (1,815 ha, tropical forest and savanna) in 2012 using aerial deployment of brodifacoum bait (25 ppm). Lizards were trapped for rodenticide testing. The authors reported that Pinzon giant tortoise *Chelonoidis ephippium* hatchling survival increased after rat eradication (see original paper for details).

(1) Newman D.G. (1994) Effects of a mouse, *Mus musculus*, eradication programme and habitat change on lizard populations of Mana Island, New Zealand, with special reference to McGregor's skink, *Cyclodina macgregori*. *New Zealand journal of zoology*, 21, 443–456.
(2) Towns D.R. (1994) The role of ecological restoration in the conservation of Whitaker's skink (*Cyclodina whitakeri*), a rare New Zealand lizard (Lacertilia: Scincidae). *New Zealand Journal of Zoology*, 21, 457–471.
(3) Risbey D.A., Calver M.C., Short J., Bradley J.S. & Wright I.W. (2000) The impact of cats and foxes on the small vertebrate fauna of Heirisson Prong, Western Australia. II. A field experiment. *Wildlife Research*, 27, 223–235.
(4) Meier G. (2003) *Eradication of invasive rats on Sangalaki-Island, East-Kalimantan – part of a project for marine turtle conservation.* InGrip-Consulting & Animal Control report, Germany.
(5) Olsson M., Wapstra E., Swan G., Snaith E., Clarke R. & Madsen T. (2005) Effects of long-term fox baiting on species composition and abundance in an Australian lizard community. *Austral Ecology*, 30, 899–905.
(6) Parrish R. (2005) Pacific rat *Rattus exulans* eradication by poison-baiting from the Chickens Islands, New Zealand. *Conservation Evidence*, 2, 74–75.
(7) Phillips R.B., Cooke B.D., Campbell K., Carrion V., Marouez C. & Snell H.L. (2005) Eradicating feral cats to protect Galapagos land iguanas: methods and strategies. *Pacific Conservation Biology*, 11, 257–267.
(8) Daltry J. (2006) The effect of black rat *Rattus rattus* control on the population of the Antiguan racer snake *Alsophis antiguae* on Great Bird Island, Antigua. *Conservation Evidence*, 3, 30–32.
(9) Rodríguez C., Torres R. & Drummond H. (2006) Eradicating introduced mammals from a forested tropical island. *Biological Conservation*, 130, 98–105.
(10) Norbury G., van den Munckhof M., Neitzel S., Hutcheon A., Reardon J. & Ludwig K. (2014) Impacts of invasive house mice on post-release survival of translocated lizards. *New Zealand Journal of Ecology*, 322–327.
(11) Hayes W.K., Cyril Jr S., Crutchfield T., Wasilewski J.A., Rothfus T.A. & Carter R.L. (2016) Conservation of the endangered San Salvador rock iguanas (*Cyclura rileyi rileyi*): population estimation, invasive species control, translocation, and headstarting. *Herpetological Conservation and Biology*, 11, 90–105.
(12) Rueda D., Campbell K.J., Fisher P., Cunninghame F. & Ponder J.B. (2016) Biologically significant residual persistence of brodifacoum in reptiles following invasive rodent eradication, Galapagos Islands, Ecuador. *Conservation Evidence*, 13, 38–38.

Crocodilians

- We found no studies that evaluated the effects of removing or controlling predators using lethal controls on crocodilian populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Tuatara

- **One study** evaluated the effects of removing or controlling predators using lethal controls on tuatara populations. This study was in New Zealand[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Abundance (1 study):** One replicated, controlled, before-and-after study in New Zealand[1] found that after eradicating Pacific rats the abundance of tuatara was higher on islands where rats were eradicated than on islands where some rats remained, and that the percentage of total tuatara that were juveniles increased.

BEHAVIOUR (0 STUDIES)

A replicated, controlled, before-and-after study in 1979–2005 on four coastal forest-covered pacific islands, New Zealand (1) found that eradicating Pacific rats *Rattus exultans* using rodenticides increased the population density of tuatara *Sphenodon punctatus* and increased the proportion of juveniles. On the three rat free islands, 162 tuatara were found over a total area of 5 ha (1–2 ha/island), compared to 44 tuatara found on the island with rats over a 39 ha area (no statistical tests were carried out). The percentage of juvenile tuatara increased after rats were eradicated on three islands (5–43%) compared to before they were eradicated (0–9%), whereas the proportion of juveniles remained at 0% on an island without rat eradication over 21 years of monitoring. Smaller tuatara were observed more frequently and in a greater range of size classes after rat eradication (see original paper for details). Rats were managed on Whatupuke Island (eradicated: 1993; 102 ha), Lady Alice Island (eradicated: 1994; 155 ha), and Coppermine Island (heavily controlled: 1992–1993; eradicated: 1997; 80 ha) using rodenticide (brodifacoum, aerial deployments except Coppermine Island in 1992–1993 when rodenticide blocks were placed on the ground). Rats were not eradicated from Taranga Island (500 ha). Tuatara were monitored on all islands at night using spotlight searches before and after rat eradication (4.5–8.5 years after) and on Taranga island in 1984, 2000, and 2005.

(1) Towns D.R., Parrish G.R., Tyrrell C.L., Ussher G.T., Cree A., Newman D.G., Whitaker A.H. & Westbrooke I. (2007) Responses of tuatara *Sphenodon punctatus* to removal of introduced Pacific rats from islands. *Conservation Biology*, 21, 1021–1031.

9.2. Remove or control predators by relocating them

- **Two studies** evaluated the effects on reptile populations of removing or controlling predators by relocating them. Both studies were in the USA[1,2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Abundance (1 study):** One before-and-after study in the USA[1] found that after raccoons were live trapped and relocated, the number of freshwater turtle hatchlings increased for 2–3 years, then decreased again after 3–4 years.
- **Reproductive success (2 studies):** One of two studies (including one replicated, controlled study) in the USA[1,2] found that within a fenced area where predators were removed by both relocating and lethal controls, fewer gopher tortoise nests were predated than outside the fenced area where predators were not removed. The other study[1] found that after raccoons were live trapped and relocated, predation of freshwater turtle nests decreased for 2–3 years, then increased again after 3–4 years.
- **Survival (1 study):** One replicated, controlled study in the USA[2] found that within a fenced area where predators were removed by both relocating and lethal controls, survival of gopher tortoise hatchlings was higher than outside the fenced area where predators were not removed.

BEHAVIOUR (0 STUDIES)

Background

Predators can drive declines or local extinctions of vulnerable reptile species. Non-native predators may be a particular problem for native reptiles that lack sufficient predator avoidance behaviours. Native predators can also threaten populations of reptiles that persist in low numbers. Predator control may attract opposition on animal welfare grounds, and therefore relocating predators may be a more favourable option in some circumstances.

Se also: *Remove or control predators using fencing and/or aerial nets* and *Remove or control predators using lethal controls.*

A before-and-after study in 1978–1982 around ponds and sand dunes in Iowa, USA (1) found that following removal of racoons *Procyon lotor*, the number of turtle hatchlings increased, and nest predation decreased for the first few years. Results were not tested statistically. Nest predation decreased for two years following racoon removal (before removal: 18 nests destroyed; after removal: 5 and 4 nests destroyed), but increased again 3–4 years after removal (21 and 28 nests destroyed). Abundance of hatchlings increased for three years (before removal: 15 hatchlings; after removal: 75, 80 and 74 hatchlings), but then decreased four years after removal (30 hatchlings). The most abundant turtle species was the yellow mud turtle *Kinosternon flavescens* (167 hatchlings seen in total). Raccoons were live trapped during 1979 and relocated to a site 24 km from the study site. Surveys for hatchlings and destroyed nests were conducted in 1978–1982. Turtle nesting areas were monitored twice/week and hatchling turtles were sampled using drift fences placed between ponds and known nesting areas 3–30 m from water and pitfalls.

A replicated, controlled study in 2002–2005 in a pine forest in Georgia, USA (2) found that removal of predators using relocations and lethal controls from fenced exclosures resulted in higher survival of gopher tortoise *Gopherus polyphemus* nests and hatchlings compared to areas with no fencing or predator removal. The effects of predator removal (both relocations and lethal controls) and fencing cannot be separated. Survival was higher inside fenced areas with predator removal compared to outside for both nests (Fenced: 52 of 78, 66% survived; Un-fenced: 26 of 73, 35%) and hatchlings (Fenced: 74% survived for 1 year; Un-fenced: 38%). In 2002–2003, four plots (40 ha) were randomly selected and enclosed in 1 m high mesh fence with electrical wires at the top and bottom. A further four plots were left un-fenced. In 2002–2003, all mammalian predators within the exclosures were live-trapped and removed, and in 2003–2005, further trapping of meso-predators was conducted. Predators that re-entered exclosures were euthanized. In May–June 2003–2005, all tortoise burrows were searched for nests, and all active nests were monitored 1–2 times/week up to 110 days. In 2004, forty hatchlings from 13 different nests were fitted with radio transmitters and monitored for up to a year.

(1) Christiansen J.L. & Gallaway B.J. (1984) Raccoon removal, nesting success, and hatchling emergence in Iowa turtles with special reference to *Kinosternon flavescens* (Kinosternidae). *The Southwestern Naturalist*, 29, 343–348.

(2) Smith L.L., Steen D.A., Conner L.M. & Rutledge J.C. (2013) Effects of predator exclusion on nest and hatching survival in the gopher tortoise. *The Journal of Wildlife Management*, 77, 352–358.

9.3. Remove or control predators using fencing and/or aerial nets

- **Ten studies** evaluated the effects on reptile populations of removing or controlling predators using fencing and/or aerial nets. Five studies were in Australia[1-3,5,10], two were in each of the USA[7,9] and New Zealand[4,8] and one was in Spain[6].

COMMUNITY RESPONSE (1 STUDY)

- **Richness/diversity (1 study):** One controlled study in Australia[3] found mixed effects of fencing in combination with removal of invasive mammals on reptile species richness.

POPULATION RESPONSE (10 STUDIES)

- **Abundance (5 studies):** Three of four studies (including one paired sites, controlled, before-and-after study) in Australia[1-3,5] found mixed effects of fencing[5] or fencing and removal of invasive mammals[1,3] on the abundance of reptiles. The other study[2] found that small lizards were more abundant inside fenced areas than outside fenced areas. This study[2] also found mixed effects of fencing on the abundance of skinks and geckos. One replicated, controlled, before-and-after study in Australia[10] found that in areas with fencing the abundance of reptiles increased more over time than in areas with no fencing.
- **Reproductive success (2 studies):** One of two replicated, controlled studies (including one randomized study) in the USA[7] and Spain[6] found that in areas with fencing in combination with predator removal[7], gopher tortoise nests[7] were predated less frequently than in areas with no corrals or fencing with predator removal. The other

study[6] found mixed effects of fencing on predation of artificial western Hermann's tortoise nests.

- **Survival (4 studies):** Two of three studies (including one replicated, randomized, controlled study) in New Zealand[4,8] and the USA[7] found that in areas with fencing in combination with predator removal, more gopher tortoise hatchlings survived for a year than in areas with no fencing or predator removal[7] or survival of captive-bred Otago skinks released into an enclosure was higher when mice had been eradicated compared to when skinks were released in the presence of mice[8]. The other study[4] found that use of predator exclosure fences did not result in increased survival of McCann's skink compared to areas without exclosures. One replicated, randomized, controlled study in the USA[9] found that in enclosures designed to exclude small mammals with additional fencing and overhead netting, a similar number of gopher tortoise hatchlings were predated by vertebrate predators compared to in unmodified enclosures.

BEHAVIOUR (0 STUDIES)

Background

Predators can drive declines or local extinctions of vulnerable reptile species. Non-native predators may be a particular problem for native reptiles that lack sufficient predator avoidance behaviours. Native predators can also threaten populations of reptiles that persist in low numbers. Predator control may be impractical to sustain on a sufficient scale or may attract opposition on animal welfare grounds. Fencing, including electric fencing, may be a viable or more effective alternative in some situations.

Studies included here are those that use fencing to exclude predators from large areas, often large enough to encompass the home ranges of the reptiles they seek to protect. For studies that use artificial covers to protect individual reptile nests, see *Protect nests and nesting sites from predation using artificial nest covers*.

See also: *Remove or control predators using lethal controls* and *Remove or control predators by relocating them*.

A controlled, before-and-after study in 1990–1994 in mixed heath and dune habitat in Western Australia, Australia (1) found that a fenced area with cat *Felis catus* and fox *Vulpes vulpes* control had similar numbers of geckos and skinks but fewer dragon lizards than unfenced areas with an without fox control. The effects of fencing and predator control cannot be separated. The number of geckos and skinks were similar in the fenced area (with fox and cat control) and unfenced (fox control or no control) area (geckos: 1–2 individuals/trap grid; skinks: 2 individuals/trap grid). Dragon lizard numbers were lower in the fenced area with fox and cat control (fox and cat control: 2 individuals/trap grid) compared to unfenced areas with fox control (fox control only: 5 individuals/trap grid) and unfenced, no control areas (no predator control: 7 individuals/trap grid). In areas with predator control, there was no clear change in reptile numbers from before control began (0–24 individuals/group/year) compared to the three years after control began (0–12 individuals/group/year). In 1991, a mainland peninsula was divided into three areas: one area (12 km^2) where cats and foxes were controlled (using electrified fencing, poison baiting, or secondary poisoning by poisoning European rabbits *Oryctolagus cuniculus*, trapping or shooting); one area (120–200 km^2) where foxes were controlled by baiting but cats were not targeted; and one area where no control occurred. Reptiles were monitored with six pitfall-trap and drift fence grids in each area (18 in total). Each grid had eight pitfall traps, 30–50 m apart. Sampling was conducted over three consecutive days in March–April and June–July in 1990–1994 in predator control areas and 1992–1994 in the area without predator control. Reptiles captured included dragon lizards, skinks, geckos, snakes, and a species of monitor lizard and blind snake but only species that could be toe clipped (dragon lizards, skinks and geckos) were included in analysis.

A replicated, site comparison study in 1999 in a site of semi-arid shrubland, grasses and sparse woody plants in New South Wales, Australia (2) found that areas within exclosure fences with ongoing fox *Canis vulpes* control had a higher abundance of small lizard species compared to areas outside the fences. Average captures of all lizard species were higher within the exclosure fences (28 lizards of 11 species; 0.9/trap) compared to outside the fences (13 lizards of 8 species; 0.5/trap) (differences in richness were not tested for statistical significance). The same was true for skinks (inside: 1.2/trap; outside: 0.3/trap). Numbers of geckos was similar inside fences (0.7/trap) and outside (0.7/trap) An area of natural habitat (400 x 200 m) was fenced off in the 1980s as part of a species reintroduction. Poison baiting for foxes was ongoing in the area. Eight pens were established (100 x 100 m in a 4 x 2 design) using chicken wire and an electrified wire. In October 1999, reptile trapping occurred inside and outside exclosures with four lines of pitfall traps (8–14 m long, buckets every 2 m).

A controlled study in 1998–2005 in a site of dunes and shrubland in South Australia, Australia (3) found that in a fenced area where invasive cats *Felis catus*, foxes *Vulpes vulpes* and European rabbits *Oryctolagus cuninculus* were removed, reptile abundance and species richness were similar for three years, then in the following five years, abundance was lower compared to outside the fenced area and richness was higher in a fenced area where native mammals had not been reintroduced. During the first three years (1998–2000), reptile abundance and species richness were similar inside and outside the fenced area (native mammals reintroduced to fenced area in 1999). In the following five years (2001–2005), the abundance of reptiles was lower inside an expanded fenced area (one area with and one without native mammals) than outside, and richness was higher in one fenced area (no native mammals) than the other fenced area (with mammals) and outside the fence (data reported as statistical model results). A netting fence was constructed in 1997 and all rabbits, cats and foxes were removed. In 1999, locally extinct small mammals were reintroduced to the fenced area. The fenced area was expanded four times in 1999–2005, and one area received no native small mammals. In 1998, twenty-four trapping sites (12 inside the fence, 12 outside) were established. In 1999, six “outside” sites became “inside” sites as the fenced area expanded, and five new “outside” sites were established. Sites were trapped for four nights (6 pitfall traps, and 10 m drift fence) in April 1998–2000 and February 2001–2005.

A randomized, controlled, before-and-after study in 2004–2006 on a coastal dune site on South Island, New Zealand (4) found that use of predator exclosure fences did not result in increased survival of McCann’s skink *Oligosoma maccanni* compared to when no exclosure fencing was used. Average change in skink survival before and after installation of exclosure fences did not differ between sites with exclosures (survival changed by 1%) and sites without exclosures (survival changed by -1%). Four sites each were assigned to one of four treatments: exclosure fences (25 x 25 m area, 1 m high chicken wire fence, bird netting on top), exclosure fence and artificial refuges (32 refuges/site); artificial refuges only; and no treatment. Skinks were sampled annually using 4-day pitfall trapping sessions in February and March 2004–2006 with fencing and refuges placed into randomly allocated grids immediately before the second year.

A paired sites, controlled, before-and-after study in 1993–1996 and 2007 in chenopod scrubland in South Australia, Australia (5) found that fencing to exclude predators and herbivores had mixed effects on different reptile species and species groups. One gecko species increased and two geckos decreased in abundance after exclusion fencing was added, compared to before when the same plots were grazed (knob-tailed gecko *Nephrurus levis* after fencing: 3.3 individuals/plot vs. grazed: 0.3–0.5 individuals/plot; tessellated gecko *Diplodactylus tessellatus* 0.0 vs. 1.3–1.8; variable fat-tailed gecko *Diplodactylus conspicullatus* 0.4 vs. 1.5–1.9). See paper for details of other species responses. Five grazed sites and four paired sites of differing grazing pressure were set out in 1993 (low intensity grazing: <12 cattle dung/ha; medium: 12–100; high: >120). Following the initial four years of the study, three of the eight grazing pressure sites were fenced to exclude cattle and predators. Reptiles were sampled for 10 days in summer from 1993–1996 and 2007 using 300 mm long flymesh drift fences with 13 unbaited pitfall traps (500 mm deep x 150 mm wide, 8 m apart). Lizards were marked by toe clips.

A replicated, controlled study in 2009 in open shrubland in Catalonia, Spain (6) found that fencing nesting sites reduced predation by some species on artificial western Hermann's tortoise *Testudo hermanni hermanni* nests and increased the time until predation occurred in one of two trials. In a first trial, artificial Hermann's tortoise nests in fenced areas survived longer until depredation (15 days) compared to artificial tortoise nests in unfenced areas (<2 days). In a second trial one month later, all fenced and unfenced nests were depredated within three days. Authors report that fencing did not prevent predation by beech martens *Martes foina,* but other predators in the area (wild boar *Sus scrofa*, red fox *Vulpes vulpes*, common genets *Genetta genetta* and European badgers *Meles meles*) were successfully excluded (see original paper for details). Predation of artificial tortoise nests (three buried quail *Corturnix coturnix* eggs) was monitored in a nature reserve in sixteen 100 m^2 plots which had been cleared to 3% of shrub cover using pruning shears. Half of the plots were enclosed with a mesh fence (200 cm high). In September and again in October 2009, one artificial nest was placed in the centre of each plot. Predation was monitored by trail cameras and visual signs.

A replicated, randomized, controlled study in 2002–2005 in a pine forest in Georgia, USA (7) found that constructing fences to exclude predators along with predator removal resulted in higher survival of gopher tortoise *Gopherus polyphemus* nests and hatchlings compared to areas with no fencing or predator removal. Survival was higher inside fenced areas compared to outside for both nests (fenced: 52 of 78, 66% survived; unfenced: 26 of 73, 35%) and hatchlings (fenced: 74% survived for 1 year; unfenced: 38%). In 2002–2003, four plots (40 ha) were randomly selected and enclosed in 1.1m high mesh fence with electrical wires at the top and bottom. A further four plots were left unfenced. In 2002–2003, all mammalian predators within the exclosures were live-trapped and removed, and in 2003–2005, further trapping of predators was conducted. Predators that re-entered exclosures were euthanized. In May–June 2003–2005, all tortoise burrows were searched for nests, and active nests were monitored 1–2 times/week for up to 110 days. In 2004, forty hatchlings from 13 different nests were fitted with radio transmitters and monitored for up to a year.

A study in 2009–2012 in an area of mixed shrub and grassland in Otago, New Zealand (8) found that survival of captive-bred Otago skinks *Oligosoma otagense* released into an enclosure was higher for those released when house mice *Mus musculus* had been eradicated compared to when skinks were released in the presence of mice. Authors reported that post-release survival was higher for skinks released with no mice present (44%) compared to survival of skinks released just prior to reinvasion by mice (15%; see paper for details). Survival of established skinks (2 years after their release) after the mouse reinvasion was higher (91%) than for newly released skinks in the presence of mice (17%). In 2009, a 0.3 ha area was enclosed within a mammal resistant fence (1.9 m high) and over a six month period, all mammals inside the enclosure were eradicated using a range of baited traps. After eradication, 12 captive-bred adult skinks were released in the enclosure following eight weeks in quarantine. In 2011, an additional 16 skinks were quarantined and released. House mice reinvaded during 2012 and were again eradicated using live capture traps and poison bait stations. In 2009–2012, starting 7–10 days after release, skinks were monitored every 15 days by a walking survey of the enclosure.

A replicated, randomized, controlled study in 2014–2015 in mixed forest and agricultural land in Georgia, USA (9) found that using fencing and overhead netting to control vertebrate predators (as well as nest cage covers) did not reduce predation of gopher tortoise hatchlings *Gopherus polyphemus*. Gopher tortoise hatchling predation by vertebrate predators in enclosures with overhead netting was the same (3 individuals) as in enclosures with no netting. Enclosures with overhead netting had fewer signs of vertebrate predators (mammals, birds and snakes) compared to those without (signs included raccoon *Procyon lotor* digging and tracks, no data provided). In May–June 2014, wild-laid gopher tortoise nests were relocated to eight fenced 0.2 ha enclosures (four nests/enclosure, two eggs/nest, 64 total eggs). All nests were covered with cloth cages (30 x 30 x 12 cm). Four of eight enclosures were covered with game farm netting and UV twine to exclude aerial and terrestrial vertebrate predators (mammals, birds and snakes). Four enclosures (two with overhead netting; two without netting) were also treated with insecticide to reduce fire ant numbers. Nests were monitored weekly until two weeks before expected emergence, daily thereafter and excavated after 120 days. Hatchlings were radio tracked (16 individuals each from insecticide-treated and untreated enclosures) from August 2014 to March 2015.

A replicated, controlled, before-and-after study in 2013–2015 in tropical savanna in the Northern Territory, Australia (10) found that erecting fencing to exclude feral cats *Felis catus* (and potentially other carnivores and herbivores) combined with fire suppression increased reptile abundance over time, but effects on reptile species richness were inconclusive. Average reptile abundance doubled over two years in plots with exclusion fencing and fire suppression (2013: 0.3 reptiles/plot; 2015: 0.7 reptiles/plot; results standardised by sampling effort), compared to plots without fencing (2013: 0.6 reptiles/plot; 2015: 0.5 reptiles/plot; results standardised by sampling effort). The effects of fencing and/or fire suppression on species richness was inconclusive (see original paper for details). Cat density in the study area was 0.2 cats/km^2. Cats were detected at all non-fenced plots during the study. Only one cat was found and removed from a fenced plot (within one week of fence completion). Data were collected from six plots (64 ha) with two each treated with: exclusion fencing and fire suppression; no exclusion fencing but fire suppression; and no exclusion fencing or fire suppression. Exclusion fences (installed December 2013) were 1,800 mm high with a curved floppy section 450 mm at the top of the fence above ground and 550 mm below ground. Fire suppression included 8 m wide firebreaks, early dry season fuel reduction burning around external perimeters, and active fire suppression inside the plots. Reptiles were monitored seasonally (March-April, June-July, October-November) in six transects/plot using drift fences and pitfall traps in 2013–2015. Cats were monitored using camera traps. Abundance of other carnivores and herbivores in/around the study site was not monitored.

(1) Risbey D.A., Calver M.C., Short J., Bradley J.S. & Wright I.W. (2000) The impact of cats and foxes on the small vertebrate fauna of Heirisson Prong, Western Australia. II. A field experiment. *Wildlife Research*, 27, 223–235.

(2) Olsson M., Wapstra E., Swan G., Snaith E., Clarke R., Madsen T. (2005) Effects of long-term fox baiting on species composition and abundance in an Australian lizard community. *Austral Ecology*, 30, 899–905.

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(5) Read J.L. & Cunningham R. (2010) Relative impacts of cattle grazing and feral animal on an Australian arid zone reptile and small mammal assemblage. *Austral Ecology*, 35, 314–324.

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(7) Smith L.L., Steen D.A., Conner L.M. & Rutledge J.C. (2013) Effects of predator exclusion on nest and hatchling survival in the gopher tortoise. *The Journal of Wildlife Management*, 77, 352–358.

(8) Norbury G., van den Munckhof M., Neitzel S., Hutcheon A., Reardon J. & Ludwig K. (2014) Impacts of invasive house mice on post-release survival of translocated lizards. *New Zealand Journal of Ecology*, 322–327.

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9.4. Use collar-mounted devices to reduce predation by domestic animals

- **Two studies** evaluated the effects of using collar-mounted devices to reduce predation by domestic animals on reptile populations. Both studies were in Australia[1,2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Survival (2 studies):** One of two replicated, randomized studies (including one before-and-after and one controlled study) in Australia[1,2] found that cats wearing collar mounted neoprene bibs, with or without a bell, caught a similar number of combined reptiles and amphibians compared to cats not wearing them[1]. The other study[2] found that cats wearing collar mounted ruffs brought home fewer combined reptiles and amphibians than cats not wearing them.

BEHAVIOUR (0 STUDIES)

Background

Domestic animals can predate a range of wild animals, with domestic cats *Felis catus* a potentially significant predator (Woods *et al*. 2003; Cecchetti *et al*. 2020). Various measures have been suggested, or are used, to try to reduce this predation, including a range of deterrents or warnings attached to collars that are worn by cats.

Woods M., Mcdonald R. & Harris S. (2003) Predation of wildlife by domestic cats *Felis catus* in Great Britain. *Mammal Review,* 33, 174–188.

Cecchetti M., Crowley S.L. & McDonald R.A. (2020) Drivers and facilitators of hunting behaviour in domestic cats and options for management. *Mammal Review*. 51, 307–322.

A replicated, randomized, before-and-after study in 2005 in urban areas in Western Australia, Australia (1) found that putting a 'CatBib™' on domestic cats *Felis catus* to inhibit pouncing, with or without a bell, did not decrease capture rates of reptiles and amphibians (combined). There was no difference in the number of cats that caught reptiles and amphibians when they wore the 'pounce protector' (10/56 cats) compared to when the same cats did not wear one (15/56 cats). The number of reptiles and amphibians captured was similar when wearing the protector (0.5/cat, 29 individuals) compared to when not wearing one (0.7/cat, 38 individuals). Changing the colour of the pounce protector did not reduce capture rates (teal: 2–3 reptiles/amphibians caught; purple: 2–4 reptiles/amphibians). Adding a bell to the pounce protector did not reduce capture rates (with bell: 1–4 reptiles/amphibians caught; no bell: 2–3 reptiles/amphibians). Reptiles caught included native skinks, geckos and lizards. The CatBib™ pounce protector is a neoprene flap that hangs from a collar in front of a cat's front legs, acting either as a visual warning or as a barrier to pouncing. Cats (male = 34, female = 28) were randomly allocated to one of four treatments: wearing a teal-coloured pounce protector, a teal pounce protector with a bell, a purple pounce protector, or a purple pounce protector with a bell. Cat owners monitored dead and live prey caught by cats for six weeks (three weeks each with and without the device) in November–December 2005. Half of the cats in each treatment were monitored wearing the device first followed by no device and the other half were monitored without the device first.

A replicated, randomized, controlled study in 2012–2014 in suburbs in Western Australia, Australia (2) found that cats *Felis catus* wearing ruff Birdsbesafe® collars brought home fewer reptile and amphibian prey (combined) compared to uncollared cats. Cats wearing Birdbesafe® collars brought home fewer reptiles and amphibians (red: 4 individuals, yellow: 2, rainbow: 1–26) than cats without collars (9, 8, 24–31). Collar colour did not significantly affect the reduction in reptiles and amphibians brought home, although overall captures of prey vertebrates with full colour vision (reptiles, amphibians and birds combined) were more reduced when rainbow-coloured collars were worn compared to vertebrates with limited colour vision (mammals). The effectiveness of red, yellow and rainbow-patterned Birdsbesafe® collars (5 cm wide ruff fitted over standard cat collars) were tested in 2012–2014 by randomly assigning colours to cats and monitoring prey captures for three-week periods with or without the collar (2012–2013: 39 cat households, 2013–2014: 43 new cat households). Only rainbow collars were tested in 2013–2014. Cat owners recorded dead and live prey captures.

(1) Calver M., Thomas S., Bradley S. & McCutcheon H. (2007) Reducing the rate of predation on wildlife by pet cats: The efficacy and practicability of collar-mounted pounce protectors. *Biological Conservation*, 137, 341–348.
(2) Hall C.M., Fontaine J.B., Bryant K.A. & Calver M.C. (2015) Assessing the effectiveness of the Birdsbesafe® anti-predation collar cover in reducing predation on wildlife by pet cats in Western Australia. *Applied Animal Behaviour Science*, 173, 40–51.

9.5. Keep domestic cats indoors at times when reptiles are most active

- We found no studies that evaluated the effects on reptile populations of keeping domestic cats indoors at times when reptiles are most active.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Domestic cats *Felis catus* are a potentially significant predator of wild reptiles (Woods *et al*. 2003; Cecchetti *et al*. 2020), and this impact may be particularly pronounced in those areas with no native feline species (Woinarski *et al.* 2018). Keeping cats indoors at times of high reptile activity may substantially reduce their impact on wild populations.

Cecchetti M., Crowley S.L. & McDonald R.A. (2020) Drivers and facilitators of hunting behaviour in domestic cats and options for management. *Mammal Review*. 51, 307–322.
Woinarski J.C.Z., Murphy B.P., Palmer R., Legge S.M., Dickman C.R., Doherty T.S., Edwards G., Nankivell A., Read J.L. & Stokeld D. (2018) How many reptiles are killed by cats in Australia? *Wildlife Research*, 45, 247–266.
Woods M., Mcdonald R. & Harris S. (2003) Predation of wildlife by domestic cats *Felis catus* in Great Britain. *Mammal Review,* 33, 174–188.

9.6. Leash or restrict domestic dog movements in reptile habitats

- We found no studies that evaluated the effects on reptile populations of leashing or restricting domestic dog movements in reptile habitats.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Domestic dogs *Canis lupus familiaris* are known to harass and predate on reptiles or their eggs, including lizards, turtles and snakes (Ruiz-Izaguirre *et al.* 2015, Weston & Stankowich 2013). Leashing domesticated dogs or restricting their movements may reduce the impact they have on reptile populations.

Ruiz-Izaguirre E., van Woersem A., Eilers K.C.H., van Wieren S.E., Bosch G., Van der Zijpp A.J. & De Boer I.J.M. (2015) Roaming characteristics and feeding practices of village dogs scavenging sea-turtle nests. *Animal conservation*, 18, 146–156.

Weston M.A. & Stankowich T. (2013) Dogs as agents of disturbance. Pages 94–113 in: M.E. Gompper, (eds.) *Free-Ranging Dogs and Wildlife Conservation.* Oxford University Press.

9.7. Protect nests and nesting sites from predation using artificial nest covers

Background

Using temporary individual nest covers may be a preferred way of protecting reptile nests when predator removal is difficult (for example due to the predator's ecology), or when it is considered to be unethical (for example involving controls on endangered species), unpopular (for example charismatic or native predators) or too costly (Buzuleciu *et al.* 2015).

Due to the number of studies found, this action has been split by species group, though no studies were found for amphisbaenians.

For studies that use fencing to exclude predators from larger areas, see *Remove or control predators using fencing and/or aerial nets*.

Buzuleciu S.A., Spencer M.E. & Parker S.L. (2015) Predator exclusion cage for turtle nests: a novel design. *Chelonian Conservation and Biology*, 14, 196–201.

Sea turtles

- **Fifteen studies** evaluated the effects of protecting nests and nesting sites from predation using artificial nest covers on sea turtle populations. Six studies were in the USA[1,3,7,9,11,12], two were in each of Turkey[2,4] and Australia[13,14], and one was in each of Greece[5], Qatar[6], Indonesia[8], Cape Verde[10] and Costa Rica[15].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (15 STUDIES)

- **Reproductive success (15 studies):** Eight of 14 studies (including 10 replicated, controlled studies) in the USA[1,3,7,9,11,12], Turkey[2,4], Qatar[6], Indonesia[8], Cape Verde[10], Australia[13,14] and Costa Rica[15] found that sea turtle[1,4], loggerhead[2,9,10,11], hawksbill[6] and artificial sea turtle[7] nests with artificial covers were predated less frequently than nests with no covers. Three studies[12,13,14] found that covering sea turtle nests had mixed effects on predation, depending on predator species[12] or year[13,14]. One study[3] found that loggerhead turtle nests with artificial covers were predated more frequently than nests with no covers. One study[8] found that olive ridley turtle nests with and without artificial covers were all predated. The other study[15] found that predation attempts of green and hawksbill turtle nests with artificial covers were similar compared to nests with no cover, but that predation success was affected by the cover design. Three studies[2,4,10] also found that sea turtle[4] and loggerhead[2,10] turtle nests with artificial covers had higher hatching success than nests with no covers. One study[11] also found that loggerhead turtle nests with artificial covers had similar hatching and emergence success compared to nests with no covers. One replicated, controlled study in Greece[5] found that covering loggerhead turtle nests had mixed effects on hatching success compared to nests with no covers.

BEHAVIOUR (0 STUDIES)

A replicated, randomized, controlled study in 1993–1994 on a long sandy beach in Florida, USA (1) found that covering sea turtle nests with a wire screen resulted in lower nest predation compared to when no screen was used. Predation of screened nests (9% and 7% of 499 and 737 nests predated) was lower than for nests with no screen (31% and 24% of 231 and 379 nests predated). An additional group of nests that received both a screen and a conditioned taste aversion treatment had similar predation to both screened nests and non-screened nests (16% and 12% of 531 and 720 nests predated). In 1993–1994, a long stretch of barrier beach (37 km) was broken down into four experimental blocks, and around 2.5 km of each block was selected for a screening trial, with two-thirds of nests in the area receiving a screen (1.2 x 1.2 m wire screen, with 5.1 x 10.2 cm mesh). Screens were secured with steel rebar at each corner. Nests in the remainder of the block either received no treatment or were part of further tests of the effect of conditioned taste aversion or raccoon *Procyon lotor* removal. Two-thirds of nests in the taste aversion area were also screened. Taste aversion involved provision of chicken eggs injected with 10 mg of oral oestrogen. Turtle nests were monitored 2–4 times/month in 1993 and 2–3 times/week in 1994.

A randomized, controlled study in 1992 on a sandy beach in Turkey (2) found that protecting loggerhead turtle *Caretta caretta* nests with wire mesh grids reduced fox *Vulpes vulpes* predation and increased hatching success compared to unprotected nests. No protected nests were predated (0 of 25 nests) whereas 63% of unprotected nests experienced at least one predation event (55 of 88 nests). Protected nests had increased hatching success compared to unprotected nests (data reported as statistical model outputs). Hatching success at nine nests that were predated before protection was greater than at nests without any protection (data reported as statistical model outputs). There was evidence of fox activity (such as digging) at eight protected nests, but the predation attempts were unsuccessful. A sandy beach (4.7 km long) was patrolled in June-August 1992 for signs of turtle nesting and fox activity. In total 25 nests were selected randomly and covered with $1m^2$ mesh grids positioned 5 cm under the sand, directly above the eggs as soon as possible after laying. Nine of the 25 nests had already been predated once before the cover was put in place. Covered nests and 88 nests without covers were monitored daily for signs of further predation (also digging and defecation) and hatching success (the number of emerging hatchlings) for 30 consecutive nights.

A replicated, controlled, paired study in 1996 on a beach in Florida, USA (3) found that loggerhead *Caretta caretta* turtle nests covered with cages had higher predation rates compared to uncovered nests. Caged sea turtle nests had higher predation rates (high predation beach sections: 42–47% nests predated, low predation beach sections: 15–22%) compared to uncaged nests (3–4% and 0–3% respectively). Approximately one third of predated nests were completely destroyed whether nests were caged (11 of 29 nests) or uncaged (1 of 3 nests). Decoy cages (with no nests) were predated in high and low predation beach sections (high: 10 of 18 nests, low: 6 of 14 nests), in some cases multiple times (high: 7 of 10 nests predated >once, low: 2). Hatchling numbers from unpredated caged or uncaged nests were similar (80 hatchlings/nest) and higher than hatchling numbers from partially predated nests (50 hatchlings). Racoons *Procyon lotor* caused 88% of predation, grey foxes *Urocyon cinereoargenteus* 11% and spotted skunks *Spilogale putorius* 1%. In May–October 1996, sea turtle nests were covered with square wire cages (76 cm square, 107 cm tall, 5 x 10 cm mesh) anchored 30 cm in the sand or left uncovered in pairs in two low (40 nest pairs) and two high predation beach sections (50 nest pairs). Nest pairs were laid within 2 days of each other and in their natural position (<15 m apart, 53 pairs) or relocated to create a pair (>4 m apart, 37 pairs). Thirty-two decoy cages not covering actual nests were placed on the beach (high predation: 18 cages; low predation: 14). All nests (including decoys) were checked daily for signs of predation until October and the likely identity of the predator. Nests were excavated three days after emergence to count successfully hatched eggs.

A replicated, controlled study in 2001–2002 on two sandy beaches in southwest Turkey (4) found that covering sea turtle nests with screens resulted in fewer eggs being predated and higher overall hatching success compared to nests that were not screened. Results were not statistically tested. Fewer nests with screens were predated (0 of 54 nests) compared to unscreened nests (Dalaman beech: Of 49 nests, 20 partially predated, 13 entirely predated, 888 eggs predated; Dalyan beach: Of 40 nests, 29 predated, 2,200 eggs predated). Overall hatching success was higher for nests with screens (screened: 74%; unscreened: 54%). Beaches were searched for nests, and those at risk of predation (54 nests on Dalaman beach) were screened with a metal grid (72 x 72 cm) and a 9 cm mesh buried 20 cm deep. A further 89 nests (49 on Dalaman beach; 40 on Dalyan beach) received no screen. Nests were monitored from June–September in 2001 (Dalyan beach) and 2002 (Dalaman beach).

A replicated, controlled study in 1987–1995 on a sandy beach on Zakynthos Island, Greece (5) found that covering loggerhead turtles *Caretta caretta* nests with metal cages resulted in variable hatching success compared to nests left in situ and nests relocated to an on-beach hatchery. Hatching success for caged nests varied from 44% to 72%, compared to 56–68% for in situ nests and 51–75% for hatchery nests. Hatching success in caged nests was lower in two of six years and higher in two of six years compared to in situ nests. From 1988, nests located within 7 m of the sea and in danger of inundation were moved to a beach hatchery (77 nests) as were nests located near invasive plants which had root systems that could grow into nests. From 1990, nests located in beach areas with tourists were protected by 50 cm circular metal mesh cages buried 15 cm in the sand (88 nests). A further 313 nests were left in situ. Nests were excavated following hatchling emergence to assess hatching success.

A replicated, controlled study in 2005 on one beach in northeast Qatar (6) reported that covering hawksbill turtle *Eretmochelys imbricata* nests with plastic nets resulted in less predation by feral cats *Felis catus* and Ruppell's foxes *Vulpes rueppelli* compared to when nests were not covered. Zero of 16 nests covered with plastic nets were predated, whereas all nests that were not covered were either partially predated (6 of 31, 19%) or completely predated (25 of 31, 81%). Before plastic nets were deployed a further three nests were partially predated, and one was completely predated. In July 2005, sixteen nests were covered with plastic nets, and 35 were left uncovered. In April–September 2005, a 1.4 km stretch of beach was patrolled five times/day, and a further 1.7 km stretch was patrolled every 7–10 days. After four nests in the intensively searched stretch of beach were predated, all nests in this stretch of beach were covered with plastic nets.

A replicated, controlled study in 2010 on a sandy beach in North Carolina, USA (7) found that both metal cages and plastic screens reduced predation of artificial sea turtle nests compared to nests with no protection. Artificial nests protected with metal cages and plastic mesh were predated less by red foxes *Vulpes vulpes* (metal cage: 0 of 12 predated; plastic mesh: 0 of 12 predated) than nests with no protection (4 of 12 predated). In an additional experiment (high predator motivation), a similar number of nests protected by plastic mesh (2 of 8) were predated compared to nests protected with metal cages (0 of 8). Artificial nests consisting of five chicken eggs and scented with dilute loggerhead egg yolk mixture were buried 29 cm deep. Twelve were protected with a metal cage (122 x 61 x 61 cm, buried 30 cm deep), 12 with a plastic mesh (2.4 m^2 centred on nest), and 12 were unprotected. In an additional experiment (high predator motivation), nests consisted of bacon and rotten chicken eggs or chicken breast and bacon scraps, and eight pairs of nests were protected with a metal cage (8 nests) or plastic mesh (8 nests). Nests were checked daily for signs of predation.

A randomized, controlled study in 2009–2010 on a sandy beach in East Java, Indonesia (8) found that using artificial nest covers to protect olive ridley turtle *Lepidochelys olivacea* nests did not improve hatching success rates. All nests, including those with artificial covers and those without were predated within one week of being laid and no eggs hatched. Olive ridley turtle nests laid in May–July 2009–2010 along an 18 km stretch of sandy beach in a national park were randomly selected to be either protected by artificial nest covers (2009: 5; 2010: 5) or unprotected (2009: 6; 2010: 14). Nests were excavated to count the number of eggs and re-buried. Protected nests were covered with a 40 x 50 x 50 cm cylindrical galvanized wire cage buried 20 cm into the sand and secured with wooden stakes. All nests were temporarily covered prior to hatching to enable hatchlings to be counted. After emergence, all nests were dug up and unhatched eggs counted.

A replicated, controlled study in 2010 on a sandy beach in South Carolina, USA (9) found that using screens to cover loggerhead turtle *Caretta caretta* nests resulted in reduced predation by coyotes *Canis latrans* compared to nests with no protection. The number of predated nests was lower for screened nests (7 of 33, 21%) compared to those with no cover (6 of 10, 60%). A similar number of screened nests were predated compared to nests covered with pepper powder on the surface (2 of 10, 20%), but nests with pepper powder below the surface had similar predation as those with no treatment (5 of 10, 50%). Thirty-three nests were covered with a plastic or metal screen (1 x 1 m), and 10 were given no screen. A further 10 nests were covered with 15 ml of habanero pepper *Capsicum chinense* powder on the surface of the nest, and 10 with pepper powder below the surface. In June–July 2010, nests were monitored for complete or partial predation every 1–3 days, and a further 12 visits were made until September.

A controlled study in 2008 on a sandy beach in Boa Viste, Cape Verde (10) found that loggerhead turtle *Caretta caretta* nests reburied in mesh cages or under netting had higher hatching success and lower rates of ghost crab *Ocypode cursor* predation compared to unprotected nests. Hatching rates were higher in nests that were protected with mesh (cage: 82%; netting: 60% success) compared to nests that were not protected (33% success). Ghost crab predation rates were lowest in nests that were buried in mesh cages (4%), compared to under mesh netting (22%) or unprotected nests (55%). Turtle nests were excavated, eggs counted and reburied in the same place either inside a mesh cage (20 nests), underneath a horizontal $1m^2$ plastic mesh buried 10 cm under the surface (20 nests) or without any protection (20 nests). Nests were monitored daily until emergence. Hatchlings were counted and released from nests with protection. Hatchling tracks were counted from nests with no protection. All nests were excavated after last emergence and remaining eggs counted for analysis.

A replicated, randomized, controlled study in 2002–2007 on two sandy beaches in Georgia, USA (11) found that using plastic mesh screens to cover loggerhead turtle *Caretta caretta* nests resulted in no predation and similar hatching and emergence success compared to nests with no covers. No nests covered with plastic mesh were predated, whereas nine nests with no cover were fully predated, and nine were partially predated (result were not statistically tested). Hatching and emergence success was similar for nests with covers (hatching: 73–76%; emergence: 67–68%) and without covers (hatching: 70–80%; emergence: 67–78%). Two stretches of beach (3 and 7 km) were searched daily during May–October 2002–2007. Nests were either covered with a 1 m^2 plastic mesh screen (85 relocated to nearby dune; 75 left in situ) or received no screen (83 relocated; 137 left in situ). Nests were monitored daily for predator activity, and five days after hatchling emergence began, nests were excavated, and the numbers of hatched and unhatched eggs and live or dead hatchlings were counted.

A study in 2005–2010 on a sandy beach on an island off the coast of Florida, USA (12) found that using nest cages to cover sea turtle (loggerhead *Caretta caretta* and green *Chelonia mydas*) nests reduced predation by raccoons *Procyon lotor* but not by feral pigs *Sus scrofa*. Covering marine turtle nests with cages reduced nest predation by raccoons in five of six years (1–20% of nests predated) compared to uncaged nests (7–69% of nests predated). Cages did not prevent feral pigs from predating nests. In August 2007, 36 of 36 remaining unhatched caged nests and 14 of 14 remaining unhatched uncaged nests were predated by pigs. Caged nests took longer to be predated by pigs (20 days) compared to uncaged nests (8.5 days). Turtle nests were monitored daily throughout the nesting season on a 13 km long stretch of beach on an island (526 ha) in 2005–2010. Located nests were covered by partially buried cages (91 cm long x 91 cm wide x 76 cm tall, 5 x 10 cm wire mesh) to protect them from raccoon predation (54–159 nests/year were caged, 8–24 nests/year were uncaged). After hatching, all nests are excavated to record hatching success and predation levels. In 2007, nest predation by pigs was observed for the first time (pigs were present on the island from 2001). Pigs were eradicated from the island in 2008 but reinvaded in 2014. See 'Use lethal controls' for more details.

A replicated, controlled study in 2014–2016 at one beach in south-eastern Queensland, Australia (13) found that using aluminium cages or plastic mesh to cover loggerhead turtle *Caretta caretta* nests from yellow-spotted goanna *Varanus panoptes* predation led to lower nest predation in one of two years. In 2014–2015, predation was lower in nests covered in plastic mesh (2 out of 11) and aluminium cages (5 out of 10), compared to nests with no covering (10 out of 11). In 2015–2016, predation did not differ significantly between nests covered in plastic mesh (0 out of 15) and nests with no covering (2 out of 16). In May 2014–June 2015, ten nests were covered with aluminium cages, 11 with plastic mesh, and 11 nests had no covering. In June–July 2015–2016, fifteen nests were covered with plastic mesh and 16 nests had no covering. Aluminium or plastic covers were buried over the top of the nest at a depth of 10–20 cm. Each nest was visited daily in early December to the end of February 2014–2016 to record predation events.

A replicated, controlled study in 2005–2014 on eight mostly connected beaches in Queensland, Australia (14) found that sea turtle nests covered with mesh were predated less frequently than those not covered during five years when fox *Vulpes vulpes* control was being carried out, but a similar amount during the following five years when foxes were not controlled. In 2005–2009 when foxes were being controlled, nests covered with mesh were predated less frequently (4–28% of 18–56 nests) than those without mesh (43–100% of 7–13 nests). In 2010–2014 when foxes were not controlled, overall predation of nests was very low, and a similar number of covered (0–4% of 25–51 nests) and uncovered nests were predated (0–25% of 0–5 nests). In 2005–2014, meshing (plastic or aluminium) was used to cover all sea turtle nests that were discovered (18–56 nests/year). A number of other nests were not discovered until after hatching and so were not covered with mesh (0–13 nests/year). In 2005–2009, a total of 19 foxes were trapped and euthanized, and a number of fox dens were fumigated (number not given). No formal fox control occurred in 2010–2014, though three foxes were removed from the area for unrelated reasons. Nests were monitored continuously throughout November–April and predation of nests by foxes was recorded.

A replicated, controlled study in 2014–2015 on one sandy beach on the Caribbean coast of Costa Rica (15) found that using a screen to cover green turtle *Chelonia mydas* and hawksbill *Eretmochelys imbricata* nests resulted in a similar number of predation attempts by domestic dogs *Canis lupus familiaris* compared to when no screen was used, but the success of predation attempts varied depending on screen type. The number of predation attempts was similar for nests covered with plastic screens compared to those without screens (data reported as statistical model results). The number of successful predation attempts depended on whether a bamboo screen (16 of 31, 52%), a plastic screen (38 of 47, 81%) or no screen (31 of 31, 100%) was used to cover the nest. Fewer predation attempts were made on nests when screens were deployed just after eggs were laid (74% of nests) compared to just before hatchlings emerged (97% of nests) (number/treatment not reported), though the likelihood of successful predation did not differ (data reported as statistical model result). In March-October 2014–2015, a total of 227 nests were either covered with a plastic or bamboo screen or were left with no screen (number/treatment not reported). Screens were buried over the top of nests at a depth of 25–30 cm for green turtle nests and 10 cm for hawksbill nests. All nests were checked for predation attempts daily during the whole incubation period.

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(6) Ficetola G.F. (2008) Impacts of human activities and predators on the nest success of the hawksbill turtle, *Eretmochelys imbricata*, in the Arabian Gulf. *Chelonian Conservation and Biology*, 7, 255–257.
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(9) Lamarre-DeJesus A.S. & Griffin C.R. (2013) Use of habanero pepper powder to reduce depredation of loggerhead sea turtle nests. *Chelonian Conservation and Biology*, 12, 262–267.
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Tortoises, terrapins, side-necked & softshell turtles

- **Seven studies** evaluated the effects of protecting nests and nesting sites from predation using artificial nest covers on tortoise, terrapin, side-necked and softshell turtle populations. Five studies were in the USA[3-7] and one was in each of the Galápagos[1] and Canada[2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (7 STUDIES)

- **Reproductive success (7 studies):** Two replicated studies (including one controlled study) in the Galápagos[1] and the USA[6] found that Galápagos giant tortoise nests surrounded by rock-walled corrals[1] and bog turtle nests covered with cages[6] were predated less frequently than unprotected nests. Two replicated studies (including one randomized, controlled study) in Canada[2] and the USA[6] found that nests of painted and snapping turtles[2] and bog turtles[6] covered with cages had similar hatching success compared to nests left uncovered. One of two replicated controlled studies (including one randomized study) in Canada[2] and the USA,[4] found that painted and snapping turtle nests protected by three different cage types were predated a similar amount[2]. The other study[4] found mixed effects of different cage designs on predation rate of artificial nests at a diamondback terrapin nesting site. One replicated, before-and-after study in the USA[5] found that diamondback terrapin nests covered by a nest box with an electrified wire were predated less frequently than nests under a box with no wire. One before-and-after study in the USA[7] found that over half of eggs from bog turtle nests covered with cages in an area grazed by cattle hatched successfully. One replicated, controlled study in the USA[3] found that diamondback terrapin nests covered with cages had hatching success of 55–93%, and 83–100% of uncaged nests were predated.

BEHAVIOUR (0 STUDIES)

A replicated, site comparison study in 1964–1972 on three islands in the Galápagos, Ecuador (1) found that protecting Galápagos giant tortoise *Geochelone elephantopus* nests with rock-walled corrals reduced predation by feral pigs *Sus scrofa*. In 1964–1970, none of the giant tortoise nests (10–25/year) on one island protected using corrals were predated by pigs. In 1970–1972, one of 262 protected nests were predated by pigs compared to 23 of 29 unprotected nests. The authors reported that corrals did not prevent dogs *Canis lupus familiaris* accessing and destroying nests elsewhere. In the 1964/1965–1969/1970 nesting seasons, 10–25 giant tortoise *Geochelone elephantopus porteri* nests/year were protected from pig predation on Santa Cruz using corrals built with lava-rock walls (1.5–2 m diameter, 1 m high). Corral use was extended in the 1970/1971 and 1971/1972 nesting seasons to protect 262 nests of three subspecies of giant tortoise (*G. e. porteri, G. e. vicina, G. e. darwini*) on three islands (Santa Cruz, San Salvador and Isabela). In 1970, twenty-nine nests were not protected with corrals on Santa Cruz and hatching outcomes monitored. In 1971–1972, introduced mammals (pigs and goats *Capra hircus*) were also controlled by shooting.

A replicated, randomized, controlled study in 2010–2011 at two lakes within mixed forest in Ontario, Canada (2) found that covering painted turtle *Chrysemys picta* and snapping turtle *Chelydra serpentine* nests with one of three different cages did not affect hatching success compared to uncaged nests, and that cage design did not affect the number of predator interactions or successful predation events. For both species, hatching success was similar for caged (painted turtle: 69–79%; snapping turtle: 73–85%) and uncaged nests (painted turtle: 60%; snapping turtle: 73%). Comparisons of three cage types (uncaged nests not included) found that there was no significant difference between the number of predator interactions (above-ground cages: 14; below-ground: 16; wooden cage: 2) and successful predation attempts (above-ground cages: 3; below-ground: 1; wooden cage: 3). Nesting sites were monitored in May–June 2010 and June–July 2011. Nests were excavated and assigned to one of four treatments: above-ground wire cage (50 nests); below-ground wire cage (49 nests); above-ground wooden cage (24 nests); or no nest covering (41 nests). Wooden cages were used only in 2011. Predator interactions and successful predations were recorded throughout the nesting season and after hatchling emergence all hatchlings and unhatched eggs were counted.

A replicated, controlled study in 2014 in one brackish wetland in New York, USA (3) found that diamondback terrapin *Malaclemys terrapin* nests that were covered with metal cages had high hatching success, and nests with no cages suffered very high levels of predation. Hatching success of nests covered with cages ranged from 55–93% (number predated not given). Nests with no cages suffered high levels of predation (15 of 18 to 15 of 15, 83–100% of nests; hatching success not given). In addition, the application of pepper powder to caged nests had no effect on hatching success (pepper: 55–93%; no pepper: 78–83%). In June–July freshly laid nests were located and 11 were covered with metal mesh cages (15 cm deep). Two nests were covered with 10 g of pepper powder, and 9 received no pepper. A further 48 nests received no cages, though 15 were covered with 10 g of pepper, and 15 with 20 g of pepper. All nests were monitored daily: uncaged nests for a minimum of seven days, and caged nests until mid-November, at which point they were excavated to determine hatching success.

A replicated, controlled study in 2013 in three estuarine sites in South Carolina, USA (4) found that covering artificial turtle nests with one of three cage designs resulted in less predation by raccoons *Procyon lotor* compared to when no nest cover was used. In a comparison between three cage designs, the "birdcage" design was more effective at preventing predation (0 of 4 nests predated) than a metal cage, (2 of 4) plastic cage (4 of 4) or no cage (4 of 4). Two further trials with the "birdcage" design found that artificial nests covered with the cage were predated less than nests with no cage (cage: 0 of 8, 100% and 25 of 84, 30% predated; no cage: 8 of 8, 100% and 71 of 84, 85%). Sixteen simulated nests were created at a diamondback terrapin *Malaclemys terrapin* nesting site by digging and immediately refilling a nest-sized hole. Three cage designs were used to cover four nests each, and four nests received no cover. Two further trials tested the "birdcage" design against no nest cover (trial 1: 8 caged, 8 un-caged; trail 2: 84 caged, 84 un-caged). Artificial nests were left for 48 hours and predation attempts were recorded.

A replicated, before-and-after study in 2013–2014 on an island site between a saltmarsh and road in Georgia, USA (5) found that electrified nest boxes provided more protection for diamondback terrapin *Malaciemys terrapin* nests from predation than a nest box alone. Fewer nests laid under nest boxes with an electric wire were predated (1 of 27 nests found) compared to those under nest boxes with no wire (16 of 16 nests found). Nests laid on the artificial nest mound yielded at least 203 hatchlings. An artificial nesting mound (22.9 m long × 3.6 m wide × 1.2 m tall) was constructed using dredge material along the shoulder of an 8.7 km causeway leading to the island. On top of the mound were placed six nest boxes (3.7 x 1.2 x 0.6 m) with a ground-level 9 cm horizontal gap to allow terrapins access but to exclude predators. For 35 days from May–June 2013, one nest box was modified to include a battery-powered electric wire along the horizontal gap opening and for 26 days from June–July 2013, all six nest boxes had electric wires. The mound was excavated to find nests and hatched eggs in November 2013 and April 2014.

A replicated, controlled study in 1974–2012 in 11 wetland sites in New Jersey and Pennsylvania, USA (6) found that caged bog turtle *Glyptemys muhlenbergii* nests had lower predation rates compared to uncaged nests, but overall hatching success was not higher. Fewer eggs were predated in caged nests (6 of 97 eggs, 6%) compared to uncaged nests (82 of 161 eggs, 51%), but overall hatching success was not higher for caged nests (caged: 42 of 97 eggs, 43%; un-caged: 53 of 161 eggs, 33%). Cages of 1 cm wire mesh were installed over nests (61 cm high, 38 cm wide) and buried 8–15 cm into the ground. In June 1974–2012, twenty-seven nests in five wetlands were covered with cages, and 55 nests in 11 wetlands were left uncaged. Eggs were monitored for at least 8–9 weeks to record predation and hatching success.

A before-and-after study in 2008–2016 in wet meadow, marsh and fen habitat in New York, USA (7) found that when artificial nest covers were used to protect bog turtle *Glyptemys muhlenbergii* nests in an area that was also grazed by cattle, a higher proportion of eggs hatched compared to when there was no grazing and no nest covers were used. Results were not statistically tested. In four years when bog turtle nests were protected by artificial covers in an area grazed by cattle, overall hatching success was 52% (58 of 112 eggs hatched). When nests were not protected and there was no grazing overall hatching success over two years was 27% (4 of 15 eggs hatched). The authors reported that the nest covers protected nests from larger predators such as raccoons *Procyon lotor,* but not from smaller, burrowing predators. In 2012–2016, bog turtle nests (3–12 nests/year, 15–47 eggs/year) in a fenced wetland being grazed by cattle (5.6 ha) were protected by mesh cloth artificial nest covers (12 x 12 x 12 cm) held in place by metal pegs. In 2014, some nests were predated before covers were put in place, so 2014 results are not included here. In 2009–2010, prior to grazing being introduced, bog turtle nests (2–3 nests/year, 7–8 eggs/year) with no nest protection were monitored. Nests were located by surveying on foot in 2009–2016.

(1) MacFarland C.G., Villa J. & Toro B. (1974) The Galápagos giant tortoises (*Geochelone elephantopus*) Part II: Conservation methods. *Biological Conservation*, 6, 198–212.

(2) Riley J.L. & Litzgus J.D. (2013) Evaluation of predator-exclusion cages used in turtle conservation: cost analysis and effects on nest environment and proxies of hatchling fitness. *Wildlife Research*, 40, 499–511.

(3) Burke R.L., Vargas M. & Kanonik A. (2015) Pursuing pepper protection: habanero pepper powder does not reduce raccoon predation of terrapin nests. *Chelonian Conservation and Biology*, 14, 201–203.

(4) Buzuleciu S.A., Spencer M.E. & Parker S.L. (2015) Predator exclusion cage for turtle nests: a novel design. *Chelonian Conservation and Biology*, 14, 196–201.

(5) Quinn D.P., Kaylor S.M., Norton T.M. & Buhlmann K.A. (2015) Nesting mounds with protective boxes and an electric wire as tools to mitigate diamond-backed terrapin (*Malaclemys terrapin*) nest predation. *Herpetological Conservation & Biology*, 10, 969–977.
(6) Zappalorti R.T., Tutterow A.M., Pittman S.E. & Lovich J.E. (2017) Hatching success and predation of bog turtle (*Glyptemys muhlenbergii*) eggs in New Jersey and Pennsylvania. *Chelonian conservation and biology*, 16, 194–202.
(7) Travis K.B., Kiviat E., Tesauro J., Stickle L., Fadden M., Steckler V. & Lukas L. (2018) Grazing for bog turtle (*Glyptemys muhlenbergii*) habitat management: Case study of a New York fen. *Herpetological Conservation and Biology*, 13, 726–742.

Snakes & lizards

- We found no studies that evaluated the effects of protecting nests and nesting sites from predation using artificial nest covers on snake and lizard populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Crocodilians

- We found no studies that evaluated the effects of protecting nests and nesting sites from predation using artificial nest covers on crocodilian populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Tuatara

- We found no studies that evaluated the effects of protecting nests and nesting sites from predation using artificial nest covers on tuatara populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

9.8. Protect nests and nesting sites from predation by camouflaging nests

- **Two studies** evaluated the effects of protecting nests and nesting sites from predation by camouflaging nests on reptile populations. One study was in the USA[1] and one was in Costa Rica[2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Reproductive success (2 studies):** One replicated, controlled study in the USA[1] found that Ouachita map turtle nests that were disguised by sweeping with a broom were predated at a similar rate as unswept nests. One before-and-after, site comparison study in Costa Rica[2] found that camouflaged (details of method not provided) olive ridley turtle nests had similar hatching and emergence success compared to nests moved to a hatchery.

BEHAVIOUR (0 STUDIES)

Background

Camouflaging nests, for example by sweeping the surface substrate to remove evidence of nest laying, removes visual cues that predators and humans may use to locate nests and so may reduce incidences of predation or poaching.

A replicated, controlled study in 2013–2014 at two riverbank sites in Wisconsin, USA (1) found that sweeping the surface of Ouachita map turtle *Graptemys ouachitensis* nests or artificial nests with a broom did not reduce nest predation by racoons *Procyon lotor*. Turtle nest predation by racoons was the same for swept (16 of 16, 100% nests predated) and unswept (19 of 20, 95% nests predated) nests. Almost all artificial nests that were swept or unswept were also dug up by racoons (swept: 18 of 19, 95%; unswept: 20 of 20, 100%). Two nesting sites (112 m^2 and 157 m^2) were divided into four adjacent, alternating areas of swept (total of 16 natural and 19 artificial nests) and unswept (total of 20 natural and 20 artificial nests) nests (2 swept and 2 unswept areas/site). A three-headed broom was dragged across the surface substrate of swept areas daily from the beginning of the monitoring period until ≥7 days after the last observed nesting event during May–July 2013–2014. In addition, artificial nests were constructed by hand and made to resemble natural nests (20 nests) or swept nests (19 nests). Predation of nests by racoons was monitored with four trail cameras at each site.

A before-and-after, site comparison study in 2005–2012 on a beach in Costa Rica (2) found that camouflaging olive ridley *Lepidochelys olivacea* sea turtle nests in situ resulted in similar hatching rates to nests that were moved to an on-beach hatchery with 24-hour monitoring. Results were not statistically tested. Hatching success was similar for nests that were camouflaged (79%) or relocated to the hatchery (79%). The emergence rate of hatchlings from camouflaged nests was 71%, compared to 77% of hatchlings from fenced hatchery nests. Egg poaching reduced from 85% in 2005 to 10% of eggs in 2006–2012. Nesting activity was monitored by nightly beach patrols (4x 4 hours/night) in July/August-December in 2006–2012 (958 nests were laid, 98–177/year). In 2006–2012, nests were either relocated to a monitored on-beach hatchery (363 nests, 38%), or camouflaged (595 nests, 61%; details of camouflaging method not provided) to discourage illegal collecting. Relocated nests were randomly allocated a 1 m^2 plot in the hatchery and dug into the sand. The hatchery was monitored 24 hours a day during the nesting season. Hatchlings from both treatments were monitored on emergence and nests were excavated after hatching due dates to check hatching success.

(1) Geller G.A. (2015) A test of substrate sweeping as a strategy to reduce raccoon predation of freshwater turtle nests, with insights from supplemental artificial nests. *Chelonian Conservation and Biology*, 14, 64–72.

(2) James R. & Melero D. (2015) Nesting and conservation of the Olive Ridley sea turtle (*Lepidochelys olivacea*) in playa Drake, Osa Peninsula, Costa Rica (2006–2012). *Revista De Biologia Tropical*, 63, 117–129.

9.9. Protect nests and nesting sites from predation using visual deterrents

- **One study** evaluated the effects of protecting nests and nesting sites from predation using visual deterrents on reptile populations. This study was in Australia[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Reproductive success (1 study):** One replicated, controlled study in Australia[1] found that a similar number of loggerhead turtle nests marked with red flags were predated compared to those marked only with wooden stakes.

BEHAVIOUR (0 STUDIES)

Background
A range of visual deterrents, including flags, may be used to deter predators from approaching reptile nests and nesting sites. If successful, such deterrents could reduce incentives for carrying out lethal control of predators.

A replicated, controlled study in 2014–2015 at one beach in south-eastern Queensland, Australia (1) found that using red flags as a visual deterrent to predators of loggerhead turtle *Caretta caretta* nests did not reduce nest predation. The number of predated nests did not differ significantly between those marked with red flags (7 out of 10) and those marked with wooden stakes (10 out of 11). Yellow-spotted goannas *Varanus panoptes* were the most common predator of nests. In May 2014–June 2015, ten nests were each marked with a bright red canvas flag (30 x 40 cm) mounted on a 1.2 m high stake inserted 50 cm into the sand, 30 cm to the side of the nest. A further 11 nests were marked with only a wooden stake. All nests were visited daily in early December to the end of February 2014–2016 to record predation events.

(1) Lei J. & Booth D.T. (2017) How best to protect the nests of the endangered loggerhead turtle *Caretta caretta* from monitor lizard predation. *Chelonian Conservation and Biology*, 16, 246–249.

9.10. Protect nests and nesting sites from predation by creating new nesting sites

- **One study** evaluated the effects of protecting nests and nesting sites from predation by creating new nesting sites on reptile populations. This study was in Spain[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Reproductive success (1 study):** One replicated, controlled study in Spain[1] found that predation rate of artificial Hermann's tortoise nests in newly created nesting sites was similar to the predation rate in natural nesting sites.

BEHAVIOUR (0 STUDIES)

Background

When suitable nesting habitat is limited, reptiles may nest at relatively high densities, and clusters of nests are more vulnerable to predation (Marchand & Litvaitis 2003). Increasing the available nesting habitat may reduce predation.

For other studies that discuss the effects of creating new nesting sites see *Habitat restoration and creation.*

Marchand M.N. & Litvaitis J.A. (2003) Effects of landscape composition, habitat features, and nest distribution on predation rates of simulated turtle nests. *Biological Conservation*, 117, 243–251.

A replicated, controlled study in 2009 in open shrubland in Catalonia, Spain (1) found that clearing shrubs to create new nesting sites did not reduce predation of artificial western Hermann's tortoise *Testudo hermanni hermanni* nests. In trials with high densities of artificial nests, predation rates of artificial Hermann's tortoise nests in new nesting sites created by clearing shrubland (44% nests predated after 48 hours and 100% predated after 144 hours) were statistically similar to predation rates in existing natural nesting sites (100% nests predated after 48 hours). In second and third trials with lower densities of artificial nests, all nests in new nesting sites and natural nesting sites were depredated within 48 hours. Predation of artificial tortoise nests (three buried quail *Corturnix coturnix* eggs) was monitored in 36 square plots (of 4, 25 and 100m^2, numbers of each sized plot not provided) in a nature reserve. In 27 plots, shrubs were cut to 0–3% ground cover to represent new nesting areas (see original paper for details) and nine plots in a natural tortoise nesting area were not managed. In May 2009, nine artificial nests were placed in the centre of each managed and unmanaged plot and, in the 25 and 100 m plots, an additional nine artificial nests were placed in one corner of each plot (total 486 artificial nests). Nests were visited every two days for one week and weekly for up to a month. Predation was monitored by trail cameras and visual signs. The trial was repeated in June and August 2009, but with only one artificial nest in the centre and, where appropriate, corner of each plot.

(1) Vilardell A., Capalleras X., Budó J. & Pons P. (2012) Predator identification and effects of habitat management and fencing on depredation rates of simulated nests of an endangered population of Hermann's tortoises. *European Journal of Wildlife Research*, 58, 707–713.

9.11. Protect nests and nesting sites from predation using chemical deterrents

- **Four studies** evaluated the effects of protecting nests and nesting sites from predation using chemical deterrents on reptile populations. Two studies were in the USA[2,3] and one was in each of Spain[1] and Australia[4].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (4 STUDIES)

- **Reproductive success (4 studies):** Three of four controlled studies (including three replicated studies) in Spain[1], the USA[2,3] and Australia[4] found that a similar number of artificial Hermann's tortoise nests[1], diamondback terrapin nests[3] and loggerhead turtle nests[4] that had chemical deterrents[1], pepper powder[3] or chilli powder[4] applied were predated compared to nests with no deterrent. The other study[2] found that fewer loggerhead turtle nets that had habanero pepper powder applied to the surface were predated than nests with no pepper powder, or nest with pepper powder below the surface.

BEHAVIOUR (0 STUDIES)

Background

Various chemicals may be used to deter predators from excavating reptile nests in search of eggs. If successful, such deterrents could reduce incentives for carrying out lethal control of predators.

A controlled study in 2006 in grasslands in Catalonia, Spain (1) found that chemical deterrents did not prevent artificial Hermann's tortoise *Testudo hermanni* nests from being predated. Results were not statistically tested. Almost all artificial Hermann's tortoise nests were predated within four days whether they were protected with chemical repellent (carnivore repellent: 63 of 64, 98% nests predated, carnivore and wild boar *Sus scrofa* repellent: 78 of 80, 98%) or not (143 of 144, 99% nests predated). Unprotected nests were generally predated more quickly than protected nests. Wild boar predated 85% of nests when wild boar and carnivore repellent were used together compared to 99% of nests when carnivore repellent only was used. Other predators included common genet *Genetta genetta*, beech marten *Martes foina* and fox *Vulpes vulpes.* Artificial nests (three quail *Coturnix coturnix* eggs, buried and watered with 15 ml of diluted tortoise urine and excrement) were created in eight plots (625 m^2/plot) in a Hermann's tortoise breeding colony. In June 2006, Schwelger© carnivore repellent was distributed in four plots (25 devices/plot; 16 nests/plot). In September 2006, wild boar repellent (Stop Jabali © Hagopur GmbH) and fresh carnivore repellent were distributed to the same four plots (20 nests/plot). Nests in experimental and untreated plots were checked for predation daily for the first 15 days and weekly for up to 3 months.

A replicated, controlled study in 2010 on a sandy beach in South Carolina, USA (2) found that using habanero pepper *Capsicum chinense* powder to cover the surface of loggerhead turtle *Caretta caretta* nests resulted in reduced predation by coyotes *Canis latrans* compared to nests with pepper powder under the surface and nests with no pepper powder. The number of predated nests was lower for surface pepper treated nests (2 of 10, 20%) compared to nests with pepper under the surface of the sand (5 of 10, 50%) and nests with no pepper (6 of 10, 60%). A similar number of surface pepper treated nests were predated compared to nests covered with a screen (7 of 33, 21%). Nests were covered with 15 ml of habanero pepper powder on the surface of the nest (10 nests), below the surface and 3 cm above the eggs (10 nests) or were given no pepper powder (10 nests). A further 33 nests were covered with a plastic or metal screen (1 x 1 m). In June–July 2010, nests were monitored for complete or partial predation every 1–3 days, and a further 12 visits were made until September.

A replicated, controlled study in 2014 in one brackish wetland in New York, USA (3) found that applying habanero pepper *Capsicum chinense* powder to diamondback terrapin *Malaclemys terrapin* nests did not decrease predation compared to nests with no pepper powder. The number of predated nests was similar for those covered with pepper powder (10 g pepper: 15 of 15, 100%; 20 g pepper: 14 of 15, 93%) and those with no pepper powder (15 of 18, 83%)

(statistical significance not assessed). The number of days until predation was also not affected by pepper treatment (10 g pepper: 1.3 days; 20 g pepper: 1.8 days) or no pepper (1.4 days). In addition, nests covered with pepper and a mesh cage had similar hatching success to nests covered with just a mesh cage (all doses of pepper: 78–83%; no pepper: 55–93%). In June–July, 30 nests were covered with habanero pepper powder (10 g: 15 nests; 20 g: 15 nests) and 18 nests received no pepper powder. A further two nests received 10 g of pepper and were covered with a metal mesh cage (buried 15 cm deep), and nine were covered with cages but received no pepper. All nests were monitored daily: uncaged nests for a minimum of seven days, and caged nests until mid-November, at which point they were excavated to determine hatching success.

A replicated, controlled study in 2014–2016 at one beach in south-eastern Queensland, Australia (4) found that applying hot chilli pepper over loggerhead turtle *Caretta caretta* nests did not reduce nest predation. The number of predated nests did not differ significantly between those with chilli powder and those without chilli in 2014–2015 (chilli: 6 of 10, 60%; no chilli: 10 of 11, 91%) and 2015–2016 (chilli: 6 of 15, 40%; no chilli: 2 of 16, 13%). Yellow-spotted goannas *Varanus panoptes* were the most common predator of nests. In May 2014–June 2015, ten nests each had 40 g of hot chilli powder applied to a 0.5 x 0.5 m area over the top at a depth of 10-20 cm and a further 11 nests had no chilli applied. In 2015–2016 (months not stated), 15 nests had chilli applied and 16 nests had no chilli applied. Each nest was visited daily in early December–February 2014–2016 to record predation events.

(1) Vilardell A., Capalleras X., Budó J., Molist F. & Pons P. (2008) Test of the efficacy of two chemical repellents in the control of Hermann's tortoise nest predation. *European Journal of Wildlife Research*, 54, 745–748.

(2) Lamarre-DeJesus A.S. & Griffin C.R. (2013) Use of habanero pepper powder to reduce depredation of loggerhead sea turtle nests. *Chelonian Conservation and Biology*, 12, 262–267.

(3) Burke R.L., Vargas M. & Kanonik A. (2015) Pursuing pepper protection: habanero pepper powder does not reduce raccoon predation of terrapin nests. *Chelonian Conservation and Biology*, 14, 201–203.

(4) Lei J. & Booth D.T. (2017) How best to protect the nests of the endangered loggerhead turtle *Caretta caretta* from monitor lizard predation. *Chelonian Conservation and Biology*, 16, 246–249.

9.12. Protect nests and nesting sites from predation using conditioned taste aversion

- **One study** evaluated the effects of protecting nests and nesting sites from predation using conditioned taste aversion on reptile populations. This study was in the USA[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Reproductive success (1 study):** One replicated, randomized, controlled, before-and-after study in the USA[1] found that a similar number of loggerhead turtle nests were predated in areas of the beach where artificial nests containing unpalatable eggs were deployed (to condition taste aversion) compared to areas with no artificial nests with unpalatable eggs.

BEHAVIOUR (0 STUDIES)

Background

Conditioned taste aversion involves placing chemicals on or near either real or artificial eggs or nests. The chemicals used may be distasteful or cause sickness or other gastrointestinal discomfort, but at a dose not intended to cause long-term harm to the animal. The intention is to create an association between the eggs and the unpleasant chemical, thereby reducing predation of reptile nests.

For other uses of conditioned taste aversion, see *Use conditioned taste aversion to prevent carnivorous reptiles from eating toxic invasive cane toads*.

A replicated, randomized, controlled, before-and-after study in 1993–1994 on a long sandy beach in Florida, USA (1) found that conditioned taste aversion using artificial nests with unpalatable eggs did not reduce predation of loggerhead turtle *Caretta caretta* nests compared to areas where no taste aversion was attempted. The number of nests predated in areas with taste aversion were similar (46% and 36 % of 246 and 390 nests) to the number of nests predated with no treatment (31% and 24% of 231 and 379 nests). Additional nests in the taste aversion area that were also covered with a wire screen were predated less than nests receiving just taste aversion (16% and 12% of 531 and 720 nests), but a similar amount to nests receiving no treatment. Consumption of artificial nests were statistically similar before (42–70% of eggs eaten), during (50–60%) and after (67–70%) taste aversion treatment. In 1993–1994, a long stretch of barrier beach (37 km) was broken down into four experimental blocks, and around 2.5 km of each selected for conditioned taste aversion. Nests in the remainder of the block either received no treatment or were part of further tests of the effect of nest screening or raccoon *Procyon lotor* removal. Fifteen artificial nests were placed in each taste aversion area consisting of 10–15 chicken eggs placed on the sand surface. Egg consumption was monitored during a pre-treatment phase (8 nights, untreated eggs), a treatment phase (8–9 night, eggs injected with 10 mg oral oestrogen) and a post-treatment phase (5 nights, untreated eggs), with eggs replaced daily. Turtle nests were monitored 2–4 times/month in 1993 and 2–3 times/week in 1994.

(1) Ratnaswamy M.J., Warren R.J., Kramer M.T. & Adam M.D. (1997) Comparisons of lethal and nonlethal techniques to reduce raccoon depredation of sea turtle nests. *Journal of Wildlife Management*, 61, 368–376.

Reduce competition with other species

9.13. Remove or control non-native reptile competitors

- We found no studies that evaluated the effects of removing or controlling non-native reptile competitors on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Non-native species are a threat to native fauna worldwide, and there is growing recognition of the negative impacts that non-native reptiles have on native species and communities (Kraus 2015). Control of non-native species can be expensive and benefits may be difficult to maintain, though actions aimed at reducing their populations may be carried out on an ongoing basis for the benefit of native reptiles.

Kraus F. (2015) Impacts from invasive reptiles and amphibians. *Annual Review of Ecology, Evolution, and Systematics*, 46, 75–97.

Reduce adverse habitat alteration by other species

9.14. Remove or control non-native/invasive plants

- **Four studies** evaluated the effects of removing or controlling non-native/invasive plants on reptile populations. Two studies were in Australia[3,4] and one was in each of South Africa[1] and the USA[2].

COMMUNITY RESPONSE (1 STUDY)

- **Richness/diversity (1 study):** One replicated, randomized, controlled, before-and-after study in Australia[3] found that areas where invasive Bitou bush were sprayed with herbicide had similar reptile species richness compared to unsprayed areas.

POPULATION RESPONSE (4 STUDIES)

- **Abundance (3 studies):** Two of three replicated, controlled studies (including two randomized and two before-and-after studies) in the USA[2] and Australia[3,4] found that areas where invasive Bitou bush[3] or para grass[4] were controlled had a similar abundance of reptiles[3] and combined reptiles and amphibians[4] compared to areas with no control. One study[4] also found that the abundance of delicate skinks was lower in areas with invasive control compared to unmanaged areas. The other study[2] found that removing invasive non-native Sahara mustard had mixed effects on the abundance of Coachella Valley fringe-toed lizards and flat-tailed horned lizards.
- **Reproductive success (1 study):** One replicated, controlled, before-and-after study in South Africa[1] found that in areas where an invasive plant was removed, nesting activity by Nile crocodiles increased more than in places with no removal.

BEHAVIOUR (0 STUDIES)

Background

Invasive plants can out compete established plant species and alter habitat structure. This may alter resource availability for reptiles. Some reptile species may benefit but, for others, invasive plants may reduce available food or shelter or change the nature of the environment such that they are at increased risk of predation. Removal or control of non-native invasive plants may be carried out in an attempt to reverse these effects.

For studies describing the effect of managing vegetation more generally, see *Habitat restoration and creation – Manage vegetation using livestock grazing;*

Manage vegetation using herbicides; Manage vegetation by cutting or mowing and *Manage vegetation by hand (selective weeding).*

A replicated, controlled, before-and-after study in 1993–1997 in shoreline habitat on a lake in Kwazulu-Natal, South Africa (1) found that manual removal of the invasive plant *Chromolaena odorata* from nesting sites increased Nile crocodile *Crocodylus niloticus* successful nesting attempts over three breeding seasons. Results were not statistically tested. Known nesting sites where invasive vegetation was removed had 40% (2 out of 5 sites nested), 80% and 60% success over three breeding seasons following removal, compared to 40% nesting success before removal. Newly created nesting sites, where invasive vegetation was completely removed, had 33% (2 out of 6 sites nested), 33% and 67% success over three breeding seasons following removal, compared to 0% success before removal. Nesting success in sites where invasive vegetation was not removed was 100% (5 out of 5 sites nested), 60%, 40% and 40% over four breeding seasons. In 1993, sixteen nest sites were chosen: five known nesting sites where the invasive plant was present and manually removed from 1994; six sites newly created by manually clearing of all invasive vegetation and root stock (4 x 4 m area); and five where the invasive plant was present and was left untreated. In 1994–1997 (three breeding seasons) invasive vegetation clearing was carried out each season. In 1993–1997, all sites were monitored using foot, boat and aerial surveys in mid-December to determine use of nesting sites.

A replicated, paired sites, controlled, before-and-after study in 2002–2007 in a site of dunes and desert scrub in California, USA (2) found that manual removal of invasive non-native Sahara mustard *Brassica tournefortii* resulted in an increase in Coachella Valley fringe-toed lizard *Uma inornata* abundance but not flat-tailed horned lizard *Phrynosoma mcallii* abundance. Overall abundance of fringe-toed lizards was higher in invasive removal plots (2.5 lizards/plot) compared to plots with no removal (1.6 lizards/plot), but flat-tailed horned lizard abundance was similar in both (removal: 0.1 lizards/plot; no removal: 0.1 lizards/plot). In yearly comparisons, fringe-toed lizard abundance was higher in removal plots in one of three years during or after removal in active dunes (2nd year of removal: 6.6 lizards/plot; no removal: 3.5 lizards/plot), but not in stabilized sand fields (removal: 1.9–2.3 lizards/dune; no removal: 1.2–2.5 lizards/dune). Paired plots (10 x 100 m plots) of mustard removal and no mustard removal were established in stabilised sand fields (15 removal plots, 15 no removal plots) and active dunes (6 removal plots, 6 no removal plots). Mustard removal was carried out by hand in 2005–2006. Reptiles were surveyed at each site six times/year from May to July 2002–2007 in the morning using sightings and tracks left in the sand.

A replicated, randomized, controlled, before-and-after study in 2010–2012 in shrubland in New South Wales, Australia (3) found that spraying invasive Bitou bush *Chrysanthemoides monilifera* ssp. *Rotundata* with herbicide did not increase reptile abundance or species richness in the year after spraying. Reptile abundance and species richness was similar after shrubland was sprayed (0.4–1.0 individuals/100 m^2; 0.4–0.5 species/100 m^2, respectively) compared to before spraying (0.6 individuals/100 m^2; 0.5 species/100 m^2) and compared to sites where Bitou bush was unsprayed (0.9–1.0 individuals/100 m^2; 0.3–0.5 species/100 m^2) and unsprayed sites without Bitou bush (0.6–1.3

individuals/100 m^2; 0.3 species/100 m^2). Species composition was similar before and after spraying and between sprayed and unsprayed sites. Reptiles were surveyed in 10 sites in March–April 2010, November 2010, and February 2011. Two sites contained invasive Bitou bush and were treated with glyphosate herbicide in May–June 2010. Eight sites were not sprayed: three contained invasive Bitou bush and five did not. Where Bitou bush was present, it comprised 40% cover in a mosaic with native vegetation. Reptiles were surveyed morning and evening (15 minutes/transect) using active searches (for example, turning over logs and rocks, raking leaf litter, lifting loose bark).

A replicated, randomized, controlled study in 2004–2006 in a seasonal wetland in Queensland, Australia (4) found that overall reptile and amphibian abundances were not affected by combinations of burning and grazing to remove invasive non-native para grass *Urochloa mutica,* but that the abundance of one skink species *Lampropholis delicata* was reduced. When non-native para grass was controlled, overall reptile and amphibian abundance was similar in grazed, burned, grazed and burned and unmanaged plots (results presented as statistical model outputs) but abundance of *Lampropholis delicata* was lower in all managed plots (burned: 3 skinks/plot; grazed: 4 skinks/plot; burned and grazed: 1 skink/plot) compared to unmanaged plots (14 skinks/plot). Para grass dominated habitat in a conservation park (3,245 ha) was divided into 12 plots (200 x 300 m each) and each plot was either burned, grazed, burned and grazed, or not managed (3 plots/management type). Burning took place in August 2004, September 2005 and November 2006. Cattle grazing took place after burning in September–December 2004, October–December 2005 and November–December 2006. Stocking levels were calculated to consume 50% of the grass biomass present/plot. Reptile and frog communities were sampled four times between 2005–2007 using three pitfall/funnel trap arrays/plot (see original paper for details). Reptiles were individually marked by toe clipping prior to release.

(1) Leslie A.J. & Spotila J.R. (2001) Alien plant threatens Nile crocodile (*Crocodylus niloticus*) breeding in Lake St. Lucia, South Africa. *Biological Conservation,* 98, 347–355.
(2) Barrows C.W., Allen E.B., Brooks M.L. & Allen M.F. (2009) Effects of an invasive plant on a desert sand dune landscape. *Biological Invasions*, 11, 673–686.
(3) Martin L.J. & Murray B.R. (2013) A preliminary assessment of the response of a native reptile assemblage to spot-spraying invasive Bitou Bush with glyphosate herbicide. *Ecological Management & Restoration*, 14, 59–62.
(4) Bower D.S., Valentine L.E., Grice A.C., Hodgson L. & Schwarzkopf L. (2014) A trade-off in conservation: Weed management decreases the abundance of common reptile and frog species while restoring an invaded floodplain. *Biological Conservation*, 179, 123–128.

9.15. Remove or control invasive or problematic herbivores and seed eaters

- **Seven studies** evaluated the effects of removing or controlling invasive or problematic herbivores and seed eaters on reptile populations. Three studies were in Australia[4-6] and one study was in each of Mauritius[1], New Zealand[2], the USA[3] and the Galápagos[7].

COMMUNITY RESPONSE (2 STUDIES)

- **Richness/diversity (2 studies):** One of two studies (one site comparison study and one controlled study) in the USA[3] and Australia[4] found that areas where feral horses had been removed had higher lizard and snake species richness than sites with horses[3]. The other study[4] found mixed effects of fencing in combination with removal of invasive mammals on reptile species richness.

POPULATION RESPONSE (7 STUDIES)

- **Abundance (7 studies):** Four of seven studies (including four controlled studies) in Mauritius[1], New Zealand[2], the USA[3], Australia[4,5,6] and the Galápagos[7] found that controlling European rabbits[1,4], grey kangaroos[6] or herbivores and predators[5], in some cases using fencing[4-6], had mixed effects on the number of sightings[1] or abundance[4-6] of different reptile species. Two studies[2,7] found that when both rabbits and Pacific rats[2] or feral goats[7] were removed the abundance of lizards[2] or the percentage of giant tortoises that were juveniles[7] increased. The other study[3] found that areas where feral horses had been removed had similar lizard and snake abundance compared to sites with horses.

BEHAVIOUR (0 STUDIES)

Background

In areas occupied by non-native grazers or where domestic animals range freely over large areas, removing or controlling grazers by fencing or lethal controls may benefit some native reptiles that rely on varied ground vegetation structures.

For other studies describing the effect of ceasing or modify grazing by livestock, see *Threat: Agriculture and aquaculture – Cease livestock grazing* and *Modify grazing regime*.

A before-and-after study in 1982 and 1989 on a volcanic island in Mauritius (1) found that European rabbit *Oryctolagus cuniculus* eradication resulted in increased encounter rates of four of six reptile species. Results were not tested statistically. Daytime encounter rates increased after rabbit eradication for four species (by 0.2–2.3 individuals/hour), decreased for one species (by 2.1 individuals/hour) and stayed the same for one species (0.5 individuals/hour both years). Night-time encounter rates increased for five species (by 0.2–1.7 individuals/hour) and stayed the same for one species (0 individuals/hour in both years). For six reptile species, the total number of individuals encountered was higher following rabbit removal (37–1,363 individuals/species seen) compared to before rabbit removal (8–883 individuals/species seen), though survey effort was higher in 1989 than in 1982. In 1986, rabbits were eradicated from the island over a period of 2 months using (an unspecified) poison. Goats had been removed by progressive shooting in 1978. Three areas on the island were searched for reptiles by teams of up to seven people that thoroughly searched all vegetation. In 1982, survey effort was 59 person hours/day and 25 person hours/night, and in 1989, effort was 117 person hours/day and 49 person hours/night.

A before-and-after study in 1986–1992 on two islands near North Island, New Zealand (2) found that eradication of European rabbits *Oryctolagus cuniculus* and Pacific rats *Rattus exulans* resulted in an increase in the abundance of resident lizards. Results were not statistically tested, and effects of herbivore and predator control cannot be separated. In forest sites, lizard numbers remained stable for five years following eradication (1986–1991: 2 lizards/100 trap days) before increasing suddenly (1992–1993: 16 lizards/100 trap days). In coastal sites there was a gradual increase from the year of eradication (3 lizards/100 trap nights) to six years after eradication (70 lizards/100 trap nights). On a nearby predator free island, lizard abundance was 4 lizards/100 trap nights in forested areas and 15–60 lizards/100 trap nights in coastal areas. Rats and rabbits were eradicated in 1986–1987 from one island (rodenticide and shooting) and a nearby island was historically free of invasive mammals. In 1986–1993, lizards were counted using pitfall traps (initially 49, increased to 69 traps over 580 m^2 on removal island) that were monitored twice/year (March and November).

A site comparison study in 1998 in seven sites of sagebrush steppe in the Great Basin, USA (3) found that sites where feral horses *Equus caballus* had been removed had more lizard and snake species but similar abundances compared to grazed sites. Sites where horses had been removed had higher species richness (5 species/site) compared to sites with feral horses (2 species/site), but similar total abundance of individuals (horses removed: 11 individuals/site; horses present: 5 individuals/site). In addition, authors reported that the percentage of expected reptile species (% of those historically present) was similar for sites with and without horse removal. Ten horse-removed and nine horse-occupied plots (135 x 135 m) were chosen that had no recent fires (<15 years); were unused by cattle for at least 20 years; and were dominated by sagebrush (*Artemisia tridentala*). Only low elevation sites were included in analysis for reptiles (5 horse occupied plots; 6 horse removed plots across 7 sites). Horses were removed 10–14 years prior to the study. Sightings of reptiles within and adjacent to (≤20 m) a trapping grid (established for small mammal trapping) were recorded during May–August 1998.

A controlled study in 1998–2005 in a site of dunes and shrubland in South Australia, Australia (4) found that removing invasive European rabbits *Oryctolagus cuninculus*, cats *Felis catus* and foxes *Vulpes vulpes* within a fenced area, in combination with reintroducing native mammals, had mixed effects on reptile abundance and species richness 1–3 year and 4–8 years after fencing and removal began. Data reported on log scale or as statistical model results. During the first three years (1998–2000), reptile abundance and species richness were similar inside and outside the fenced area (native mammals reintroduced to fenced area in 1999). In the following five years (2001–2005), the abundance of reptiles was lower inside an expanded fenced area (one area with and one without native mammals) than outside, and richness was higher in one fenced area (no native mammals) than in both the other fenced area (with mammals) and outside the fence. A netting fence was constructed in 1997 and all rabbits, cats and foxes were removed. In 1999, locally extinct small mammals were reintroduced to the fenced area. The fenced area was expanded four times in 1999–2005, and one area received no native small mammals. In 1998, twenty-four trapping sites (12 inside the fence, 12 outside) were established. In 1999, six "outside" sites became "inside" sites as the fenced area expanded, and five new "outside" sites were established. Sites were trapped for four nights (6 pitfall traps, and 10 m drift fence) in April 1998–2000 and February 2001–2005.

A paired sites, controlled, before-and-after study in 1993–1996 and 2007 in chenopod scrubland in South Australia, Australia (5) found that fencing to exclude herbivores and predators had mixed effects on different reptile species and species groups. One gecko species increased and two geckos decreased in abundance after exclusion fencing was added, compared to before when the same plots were grazed (knob-tailed gecko *Nephrurus levis* after fencing: 3.3 individuals/plot vs. grazed: 0.3–0.5 individuals/plot; tessellated gecko *Diplodactylus tessellatus* 0.0 vs. 1.3–1.8; variable fat-tailed gecko *Diplodactylus conspicullatus* 0.4 vs. 1.5–1.9). See paper for details of other species responses. Five grazed sites and four paired sites of differing grazing pressure were set out in 1993 (low intensity grazing: <12 cattle dung/ha; medium: 12–100; high: >120). Following the initial four years of the study, three of the eight grazing pressure sites were fenced to exclude cattle and predators. Reptiles were sampled for 10 days in summer from 1993–1996 and 2007 using 300 mm long flymesh drift fences with 13 unbaited pitfall traps (500 mm deep x 150 mm wide, 8 m apart). Lizards were marked by toe clips.

A replicated, controlled study in 2007–2010 in two grassy woodland reserves near Canberra, Australia (6) found that fencing to reduce grey kangaroo *Macropus giganteus* grazing intensity had mixed effects on small skink abundance compared to not fencing depending on the amount of vegetation and whether coarse woody debris was added. At high vegetation density, small skink abundance increased over four years in fenced areas, but decreased in unfenced areas, whereas at medium-density vegetation the reverse was true (results reported on log scale). At low-density vegetation, small skink numbers remained stable over four years in both fenced and unfenced areas. In fenced low and medium-density vegetation sites, adding coarse woody debris (particularly 20 tonnes/ha clumped) lead to an increase in small skink abundance over time compared to when no debris was added (see paper for details). Reptiles were monitored in 96 plots (1 ha) in 24 sites across two nature reserves (4 plots/site). In October 2007, coarse woody debris was added to 72 plots (either 20 tonnes/ha evenly dispersed, 20 tonnes/ha clumped, 40 tonnes/ha dispersed and clumped) and none added to 24 plots. In December 2007, six sites were fenced to exclude kangaroos and grazing levels were classed as low (fenced: 0.4 kangaroos/ha) or high (unfenced: 2.1). Reptiles were surveyed at each site using 30-minute active searches from March to April in 2007–2010.

A controlled study in 1995–2005 on two islands in the Galápagos (7) found that removing feral goats *Capra hircus* resulted in an increase in the percentage of juvenile giant tortoises *Chelonoidis nigra vandenburghi*, whereas the percentage of juvenile giant tortoises on an island with no goat removal remained stable. With goat removal, the percentage of tortoises captured that were juveniles was higher in the second phase of goat removal (2000–2005: 24% of tortoises were juveniles) compared to the first phase (1995–1999: 5% juveniles), whereas at two locations with no goat removal juvenile numbers remained constant (1995–1999: 3% and 1%; 2000–2005: 1% and 2%). With goat removal, a total of 669 tagged tortoises were recaptured over the course of the study, and with no goat removal, 103 tortoises were recaptured. Goat removal was carried out on one island in 1995–2005. A total of 62,868 goats were removed, with around 85% of those goats being removed in the initial phase (1995–1999). No goat control was carried out on the other island. On the goat removal island, tortoises were sampled along 2–8 km long transects (placed randomly in four altitudinal zones) for 11 years; twice a year from 1995–2000 and once a year from 2001–2005, and all tortoises were individually marked. The same monitoring approach was used at two locations on the island without goat removal.

(1) North S.G., Bullock D.J. & Dulloo M.E. (1994) Changes in the vegetation and reptile populations on Round Island, Mauritius, following eradication of rabbits. *Biological Conservation*, 67, 21–28.

(2) Towns D.R. (1994) The role of ecological restoration in the conservation of Whitaker's skink (*Cyclodina whitakeri*), a rare New Zealand lizard (Lacertilia: Scincidae). *New Zealand Journal of Zoology*, 21, 457–471.

(3) Beever E.A. & Brussard P.F. (2004) Community-and landscape-level responses of reptiles and small mammals to feral-horse grazing in the Great Basin. *Journal of Arid Environments*, 59, 271–297.

(4) Moseby K.E., Hill B.M. & Read J.L. (2009) Arid Recovery—a comparison of reptile and small mammal populations inside and outside a large rabbit, cat and fox-proof exclosure in arid South Australia. *Austral Ecology*, 34, 156–169.

(5) Read J.L. & Cunningham R. (2010) Relative impacts of cattle grazing and feral animal on an Australian arid zone reptile and small mammal assemblage. *Austral Ecology*, 35, 314–324.

(6) Manning A.D., Cunningham R.B. & Lindenmayer D.B. (2013) Bringing forward the benefits of coarse woody debris in ecosystem recovery under different levels of grazing and vegetation density. *Biological Conservation*, 157, 204–214.

(7) Márquez C., Gibbs J.P., Carrión V., Naranjo S. & Llerena A. (2013) Population response of giant Galápagos tortoises to feral goat removal. *Restoration Ecology*, 21, 181–185.

Reduce adverse impacts on carnivorous reptiles of consuming poisonous non-native species

9.16. Remove or control toxic invasive amphibians (e.g. cane toads, Asian toads)

- We found no studies that evaluated the effects of removing or controlling toxic invasive amphibians on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Invasive amphibians such as the cane toad *Bufo marinus* pose a threat to predatory reptiles (including crocodiles, snakes, varanid and scincid lizards) due to their toxicity (Shine & Wiens 2010). Reductions of predatory reptiles caused by cane toads may also have knock-on effects that lead to wider changes in native reptile communities (Feit *et al.* 2020). A range of methods for controlling cane toads, including biological control, have been proposed (e.g. Shanmuganathan *et al.* 2010; Ward-Fear *et al.* 2010).

Feit B., Dempster T., Jessop T.S., Webb J.K. & Letnic M. (2020) A trophic cascade initiated by an invasive vertebrate alters the structure of native reptile communities. *Global change biology*, 26, 2829–2840.

Shanmuganathan T., Pallister J., Doody S., McCallum H., Robinson T., Sheppard A., Hardy C., Halliday D., Venables D., Voysey R., Strive T., Hinds L. & Hyatt A. (2010) Biological control of the cane toad in Australia: a review. *Animal Conservation*, 13, 16–23.

Shine R. & Wiens J.J. (2010) The ecological impact of invasive cane toads (*Bufo marinus*) in Australia. *The Quarterly review of biology*, 85, 253–291.

Ward-Fear G., Brown G.P. & Shine R. (2010) Using a native predator (the meat ant, *Iridomyrmex reburrus*) to reduce the abundance of an invasive species (the cane toad, *Bufo marinus*) in tropical Australia. *Journal of Applied Ecology*, 47, 273–280.

9.17. Use conditioned taste aversion to prevent carnivorous reptiles from eating toxic invasive cane toads

- **Two studies** evaluated the effects on reptile populations of using conditioned taste aversion to prevent carnivorous reptiles from eating toxic invasive cane toads. Both studies were in Australia[1,2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Survival (2 studies):** One of two replicated, controlled studies in Australia[1,2] found that survival of yellow-spotted goannas[2] subjected to conditioned taste aversion was higher at one of two sites[2] than those that were not treated. The other study[1] found that survival of bluetongue skinks[1] given a high dose was higher than those given a low dose, but similar to skinks receiving no dose.

BEHAVIOUR (1 STUDY)

- **Behaviour change (1 study):** One replicated, controlled study in Australia[2] found that yellow-spotted goannas subjected to conditioned taste aversion were less likely to eat cane toads than those that were not treated.

Background

Conditioned taste aversion as a conservation tool was originally developed to protect eggs and nests of birds and reptiles from predation (Maguire *et al.* 2010). In such cases, chemicals are placed on or near either real or artificial eggs or nests with the aim of teaching a predator population that eggs are distasteful or cause sickness. More recently, a novel use of conditioned taste aversion has been used in the case of native carnivorous reptiles and mammals eating the introduced and highly toxic cane toad *Rhinella marina* in Australia. In this case, conditioned taste aversion has been used with the aim of teaching the native populations that cane toads are distasteful or cause sickness and thus are not prey items. For other uses of conditioned taste aversion, see *Protect nests and nesting sites from predation using conditioned taste aversion*.

Maguire G.S., Stojanovic D. & Weston M.A. (2010) Conditioned taste aversion reduces fox depredation on model eggs on beaches. *Wildlife Research*, 36, 702–708.

A replicated, controlled study in 2010–2011 in a site of mixed bushland and agriculture in Western Australia (1) found that northern bluetongue skinks *Tiliqua scincoides intermedia* subjected to conditioned taste aversion were more likely to survive contact with invasive cane toads *Rhinella marina* when given a high dose compared with a low dose treatment, but survival after a high dose was similar to those given no dose. Survival of skinks receiving a high dose taste aversion treatment was higher (9 of 9, 100% of skinks survived) than those receiving a low dose (4 of 8, 50% survived), but similar to those receiving no dose (12 of 15, 80% survived). The high dose induced vomiting in all skinks. Skinks were located by driving slowly in the morning and late afternoon along a 14 km stretch of road between September 2010 and April 2011. Those captured were fitted with radio transmitters. At the first appearance of cane toads (December 2010), skinks were randomly allocated to a taste aversion treatment (high dose: 1.2 mg/kg, 8 M LiCl; low dose: 0.8 mg/kg) or no treatment group and skinks caught after cane toad arrival were alternately allocated to either group. All skinks received cane-toad sausage baits (high dose: 9 skinks; low does: 8; no dose: 15 skinks) and were subsequently monitored for survival.

A replicated, controlled study in 2013–2015 in two tropical floodplain sites in Western Australia, Australia (2) found that conditioned taste aversion training of yellow-spotted monitors *Varanus panoptes* using live cane toads *Rhinella marina* resulted in higher survival of goannas at one of two sites compared to those receiving no conditioning. After conditioning, goannas were less likely to eat another cane toad (1% ate a toad) compared to before conditioning (52% ate a toad). Conditioned goanna had higher survival than unconditioned goannas in the southern site (conditioned: 40% survived 400 days; unconditioned: 0% survived 200 days), but no difference was found in the northern site (conditioned: 50% survived 300 days; unconditioned: 50% survived 300 days, 20% survived 400 days). The southern site was invaded by large numbers of toads, whereas toads arrived later and in smaller numbers to the northern site. Three months prior to the toad invasion, free-ranging goannas were exposed to small live toads (greater than 25g, 30–70 mm snout-vent length) with venom squeezed out. Goannas either bit the toad (conditioned; 22 goannas) or ignored it (unconditioned; 44 goannas). Goannas were monitored in the southern (47 goannas) and northern (19 goannas) sites from November 2013 to May 2015.

(1) Price-Rees S.J., Webb J.K. & Shine R. (2013) Reducing the impact of a toxic invader by inducing taste aversion in an imperilled native reptile predator. *Animal Conservation*, 16, 386–394.

(2) Ward-Fear G., Pearson D.J., Brown G.P., Rangers B. & Shine R. (2016) Ecological immunization: in situ training of free-ranging predatory lizards reduces their vulnerability to invasive toxic prey. *Biology Letters*, 12, 20150863.

Reduce parasitism and disease

9.18. Dispose of waste from pet reptile enclosures carefully to prevent spread of disease

- We found no studies that evaluated the effects on reptile populations of disposing of waste from pet reptile enclosures carefully to prevent spread of disease.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Pet reptiles may harbour diseases that could be transferred to wild populations, and waste from pet enclosures may harbour such diseases. Adopting careful disposal practices for waste from pet reptile enclosures may reduce the potential for transmission to wild populations.

9.19. Carry out surveillance of reptiles for early treatment/action to prevent spread of disease

- We found no studies that evaluated the effects on reptile populations of carrying out surveillance of reptiles for early treatment/action to prevent spread of disease.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Carrying out surveillance of reptiles for diseases could provide an early warning system for new outbreaks and may allow preventative measures to be taken. Surveillance programmes and sampling protocols should aim to minimize disturbance to reptiles.

9.20. Sterilize equipment to prevent spread of disease

- We found no studies that evaluated the effects on reptile populations of sterilizing equipment to prevent spread of disease.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

The movement of field biologists between different sites increases the risk of spreading wildlife diseases. Precautions therefore need to be taken to reduce the risk of spreading diseases between sites and populations. This is also the case within and between captive populations. A range of methods may be used to sterilize equipment, including using disinfectants, heating and drying.

9.21. Control ectoparasites in wild reptile populations

- **One study** evaluated the effects on reptile populations of controlling ectoparasites in wild reptile populations. This study was in New Zealand[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Reproductive success (1 study):** One controlled study in New Zealand[1] found that McCann's skinks treated for mites had more successful pregnancies and produced more viable offspring than untreated skinks.

BEHAVIOUR (0 STUDIES)

Background

Although the effects of parasites, such as ticks and mites, on their hosts are often undetectable, there can be serious adverse health effects of high parasite burdens, including reduced reproductive output and increased mortality (Wall 2007). Treatments, developed primarily for domestic animals, may be administered to wild reptiles to reduce parasite burdens. The administering of such treatments, though, can be challenging.

Wall R. (2007) Ectoparasites: future challenges in a changing world. *Veterinary Parasitology*, 148, 62–74.

A controlled study in 2004 and 2007 in laboratory conditions in South Island, New Zealand (1) found that treating mites on wild-caught pregnant female McCann's skinks *Oligosoma maccanni* with vegetable oil improved pregnancy success and offspring viability. When mites were treated with vegetable oil, the majority of wild-caught pregnant female McCann's skinks gave birth successfully (22 of 30 skinks completed pregnancy successfully, 2 of 30 skinks had partially successful pregnancies), whereas when mites were not treated, most pregnancies were not successful (1 of 17 skinks had a partially successful pregnancy). Female McCann's skinks treated for mites produced more viable offspring (2.6 offspring/female), compared to when mites were not treated (0.1 offspring/female). Two weeks after initial treatment with oil, 14 of 30 female skinks showed signs of mites still being present. After 28 days (and two treatments of oil), no live mites were observed. In October 2004 and 2007, pregnant female McCann's skinks were taken from the wild and maintained in controlled temperature and lighting conditions in individual containers (2004: 17 individuals; 2007: 30 individuals; see original paper for details). In 2004, all skinks had scale mites and were not treated. In 2007, all skinks were treated for mites using sunflower oil following capture. Skinks were checked for mites and retreated with oil as necessary on the 14th day (all skinks oiled), 28th (only those skinks with raised scales were re-oiled) and 56th (no skinks were re-oiled) day following capture.

(1) Hare K.M., Hare J.R. & Cree A. (2010) Parasites, but not palpation, are associated with pregnancy failure in a captive viviparous lizard. *Herpetological Conservation and Biology*, 5, 536–570.

10. Threat: Pollution

Pollution, which can be classified as contaminants (e.g. pesticides, metals, nitrogenous compounds, pharmaceuticals, plastics, radioactive molecules, polychlorinated biphenyls and other industrial waste products), light, heat or noise, has direct and indirect consequences to reptiles. Contaminants, which can cause direct mortality and sublethal effects on reptiles (Todd *et al*. 2010), have been identified as one of the major contributors to the global decline of reptiles (Gibbons *et al*. 2000), yet reptiles remain one of the least studied vertebrate groups in ecotoxicology (Sparling *et al*. 2010, Zychowski *et al*. 2017). Light, heat and noise are even less well-studied in reptiles, but are known to impact on reproductive success, especially in sea turtles (McArthur 2004). Pollution specific to mining is discussed in *Threat: Energy production and mining*.

Gibbons J.W., Scott D.E., Ryan T.J., Buhlmann K.A., Tuberville T.D., Metts B.S., Greene J.L., Mills T., Leiden Y., Poppy S. & Winne C.T. (2000) The global decline of reptiles, déjà vu amphibians. *Bioscience,* 50, 653–666.

McArthur S. (2004) Appendix A: Turtle conservation. In: S. McArthur, R. Wilkinson, & J. Meyer (eds.) *Medicine and Surgery of Tortoises and Turtles,* Blackwell Publishing Ltd., Oxford, UK.

Sparling D.W., Linder G., Bishop C.A. & Krest S.K. (2010) Recent advancements in amphibian and reptile ecotoxicology. Pages 1–11 in: D.W. Sparling, G. Linder, C.A. Bishop & S.K. Krest (eds.) *Ecotoxicology of Amphibians and Reptiles, Second Edition*. CRC Press, Florida.

Todd B.D., Willson J.D. & Gibbons J.W. (2010) The global status of reptiles and causes of their decline. Pages 47–67 in: D.W. Sparling, G. Linder, C.A. Bishop, S.K. Krest (eds.) *Ecotoxicology of Amphibians and Reptiles, Second Edition*. CRC Press, Florida.

Zychowski G.V. & Godard-Codding C.A.J. (2017) Reptilian exposure to polycyclic aromatic hydrocarbons and associated effects. *Environmental Toxicology and Chemistry,* 36, 25–35.

General

10.1. Introduce legislation to control the use of hazardous substances

- We found no studies that evaluated the effects on reptile populations of introducing legislation to control the use of hazardous substances.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Introducing legislation to control the use of hazardous substances across a range of sectors (e.g. agriculture, manufacturing, energy production) could reduce the negative impacts on wildlife, including reptiles. Such laws exist in some countries.

10.2. Use 'bioremediating' organisms to remove or neutralize pollutants

- We found no studies that evaluated the effects on reptile populations of using 'bioremediating' organisms to remove or neutralize pollutants.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Some sources of pollution can be biologically 'remediated' by transplanting or translocating particular organisms (e.g. algae, bacteria and fungi) to the affected area (e.g. Sode *et al.* 2013, Xue *et al.* 2015). These 'bioremediating' organisms can naturally remove or neutralize pollutants and improve water quality. Transplanting or translocating such organisms to an affected area may reduce pollution levels and potential harm to reptiles.

Sode S., Bruhn A., Balsby T.J.S., Larsen M.M., Gotfredsen A. & Rasmussen M.B. (2013) Bioremediation of reject water from anaerobically digested waste water sludge with macroalgae (*Ulva lactuca, Chlorophyta*). *Bioresource Technology*, 146, 426–435.

Xue J., Yu Y., Bai Y., Wang L. & Wu Y. (2015) Marine oil-degrading microorganisms and biodegradation process of petroleum hydrocarbon in marine environments: a review. *Current Microbiology,* 71, 220–228.

10.3. Add chemicals or minerals to sediment to remove or neutralize pollutants

- We found no studies that evaluated the effects on reptile populations of adding chemicals or minerals to sediment to remove or neutralize pollutants.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Sediments within aquatic environments can accumulate pollutants over time, such as those leaching from aquaculture systems, sewage outfalls or nearby agricultural fields. Polluted sediments may negatively affect aquatic and semi-aquatic reptiles. Chemicals or minerals (e.g. coal ash, red mud and apatite) may be added to sediments to reduce or remove pollutants (e.g. Kim *et al.* 2014, Shin & Kim 2016).

Kim K., Hibino T., Yamamoto T., Hayakawa S., Mito Y., Nakamoto K. & Lee I.-C. (2014) Field experiments on remediation of coastal sediments using granulated coal ash. *Marine Pollution Bulletin,* 83, 132–137.

Shin W. & Kim Y.-K. (2016) Stabilization of heavy metal contaminated marine sediments with red mud and apatite composite. *Journal of Soils and Sediments,* 16, 726–735.

Garbage and solid waste

10.4. Limit, cease or prohibit dumping of garbage and other solid waste

- We found no studies that evaluated the effects on reptile populations of limiting, ceasing or prohibiting dumping of garbage and other solid waste.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background
Garbage and other solid waste may pose a number of threats to reptiles, including the potential for entanglement and ingestion, as well as though the introduction of harmful chemicals. Measures aimed at reducing the dumping of this waste may reduce the potential for harm to wild reptile populations.

10.5. Remove garbage and other solid waste from terrestrial, aquatic and coastal environments

- **One study** evaluated the effects of removing garbage and other solid waste from terrestrial, aquatic and coastal environments on reptile populations. This study was in the USA[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Reproductive success (1 study):** One controlled, before-and-after study in the USA[1] found that removing beach debris from one section of beach did not increase nesting success in that section.

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One controlled, before-and-after study in the USA[1] found that after the removal of beach debris from one of three beach sections, a higher percentage of both the total nests laid and failed nesting attempts occurred in that section.

Background
Garbage and other solid waste may pose a number of threats to reptiles, including the potential for entanglement and ingestion, as well as though the introduction of harmful chemicals. Removing this waste from the environment may reduce the risk of harm to wild reptile populations.

For studies that incorporate removal of garbage and solid waste as part of wider habitat restoration measures, see *Habitat restoration and creation – Whole habitat restoration*. For studies of the recovery of derelict fishing gear, see *Recover lost or discarded fishing gear*.

A controlled, before-and-after study in 2011–2014 on a beach in north-west Florida, USA (1) found that restoring a beach by removing debris (man-made and natural) increased both the percentage of total loggerhead turtle *Caretta caretta* nests laid and failed nesting attempts in the restored section, and that nesting success remained similar when debris was left in place. The percentage of total nests that were laid in the beach section cleared of debris increased after removal (27 of 84 nests, 32%) compared to before (9 of 74 nests, 12%), whereas the percentage of total nests laid in the two no-removal sections decreased in one case (after: 15%; before: 32%) and stayed the same in the other (after: 52%; before: 58%). The percentage of failed nesting attempts ('false crawls') in the beach section cleared of debris also increased after removal (45 of 131 crawls, 34%) compared to before (29 of 170 crawls, 17%), and decreased in the two no-removal sections (after: 15–50%; before: 25–58%). Nest success rate was similar after debris removal (after: 38% success; before: 24% success). The beach (5.7 km total length) was divided into three sections that initially had natural debris only (1.3 km long); man-made and natural debris (1.7 km long, 'middle'); or comparatively little debris (2.7 km long). All man-made (concrete, pipes, metal fencing) and natural (fallen trees and stumps due to erosion of an adjacent pine forest) debris were recorded (June–December 2012) and removed from the middle section only in December 2012. Nesting activity was monitored on all three beach sections daily in May–September 2011–2014 (two years before and after removal).

(1) Fujisaki I. & Lamont M.M. (2016) The effects of large beach debris on nesting sea turtles. *Journal of Experimental Marine Biology and Ecology*, 482, 33–37.

10.6. Install stormwater traps to prevent garbage from reaching rivers, coastal and marine environments

- We found no studies that evaluated the effects on reptile populations of installing stormwater traps to prevent garbage from reaching rivers, coastal and marine environments.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Garbage from urban areas can enter marine and freshwater environments in stormwaters running off land via stormwater conduits and drainage systems (Armitage & Rooseboom 2000). Garbage can negatively affect marine reptiles, and all seven species of sea turtles are known to be affected by litter in the marine environment (Kühn *et al.* 2015). Stormwater traps or grids are designed to prevent garbage from entering stormwaters and may therefore reduce the amount reaching marine and freshwater environments (Armitage 2007).

Armitage N. & Rooseboom A. (2000) The removal of urban litter from stormwater conduits and streams: Paper 1 - The quantities involved and catchment litter management options. *Water Science and Technology,* 26, 181–188.

Armitage N. (2007) The reduction of urban litter in the stormwater drains of South Africa. *Urban Water Journal,* 4, 151–172.

Kühn S., Bravo Rebolledo E.L. & van Franeker J.A. (2015) Deleterious effects of litter on marine life. Pages 75–116 in*:* M. Bergmann, L. Gutow & M. Klages (eds.) *Marine Anthropogenic Litter.* Springer International Publishing, Cham.

10.7. Use biodegradable materials to construct fishing gear to prevent entanglement of reptiles in lost or abandoned gear

- We found no studies that evaluated the effects on reptile populations of using biodegradable materials to construct fishing gear to prevent entanglement of reptiles in lost or abandoned gear.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Abandoned, lost or otherwise discarded fishing gear (or 'ghost' gear) is a major threat to aquatic reptiles. Sea turtles in particular are at risk of becoming entangled in 'ghost' gear, such as nets, lines and ropes resulting in injury or death (Stelfox *et al.* 2016). Synthetic materials used for fishing gear, such as nylon, may persist for decades leading to an accumulation of 'ghost' gear in marine and freshwater environments. Biodegradable fishing gear, which is naturally broken down by microbes or ultraviolet light, may offer an alternative to traditional materials (Kim *et al.* 2016) and help to reduce the impact of 'ghost' gear on reptiles. The degraded products of biodegradable materials (carbon dioxide, methane, water) also have no impact on marine ecosystems, unlike synthetic materials which eventually degrade into microplastics.

Kim S., Kim P., Lim J., An H. & Suuronen P. (2016) Use of biodegradable driftnets to prevent ghost fishing: physical properties and fishing performance for yellow croaker. *Animal conservation*, 19, 309–319.

Stelfox M., Hudgins J. & Sweet M. (2016) A review of ghost gear entanglement amongst marine mammals, reptiles and elasmobranchs. *Marine Pollution Bulletin,* 111, 6–17.

10.8. Prevent the loss and discard of fishing gear and related debris

- We found no studies that evaluated the effects on reptile populations of preventing the loss and discard of fishing gear and related debris.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Abandoned, lost or otherwise discarded fishing gear (or 'ghost' gear) is a major threat to aquatic reptiles. Sea turtles in particular are at risk of becoming entangled in 'ghost' gear, such as nets, lines and ropes resulting in injury or death (Stelfox *et al.* 2016).

Potential options for reducing the amount of discarded fishing gear include: offering incentives for recovering, reusing or recycling gear; equipping ports with dedicated fishing gear disposal facilities; improving methods for locating abandoned gear; establishing fishing gear registration programmes to encourage responsible use of gear; and informing fishers of the impacts of derelict fishing gear on reptiles to encourage its responsible disposal.

For studies on the recovery of fishing gear that has already been discarded see *Recover lost or discarded fishing gear*.

10.9. Recover lost or discarded fishing gear

- We found no studies that evaluated the effects on reptile populations of recovering lost or discarded fishing gear.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Abandoned, lost or otherwise discarded fishing gear (or 'ghost' gear) is a major threat to aquatic reptiles. Sea turtles in particular are at risk of becoming entangled in 'ghost' gear, such as nets, lines and ropes resulting in injury or death (Stelfox *et al.* 2016). Recovering derelict fishing gear from marine and freshwater environments may reduce the risk of mammal entanglement. However, derelict gear may be difficult to locate and retrieve. Specialist techniques may be required, such as acoustic sonar surveys, aerial surveys, or underwater diver and camera surveys (Drinkwin 2018).

For studies on the avoidance of losing or discarding gear see *Prevent the loss and discard of fishing gear and related debris*. For studies relating to the removal of other garbage see *Remove garbage and other solid waste from terrestrial, aquatic and coastal environments*.

Drinkwin J. (2018) *Methods to locate derelict fishing gear in marine waters*. A Guidance Document of the Global Ghost Gear Initiative Catalyze and Replicate Solutions Working Group, Natural Resources Consultants Inc.

Stelfox M., Hudgins J. & Sweet M. (2016) A review of ghost gear entanglement amongst marine mammals, reptiles and elasmobranchs. *Marine Pollution Bulletin*, 111, 6–17.

10.10. Remove derelict fishing gear from reptiles found entangled

- We found no studies that evaluated the effects on reptile populations of removing derelict fishing gear from reptiles found entangled.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Abandoned, lost or otherwise discarded fishing gear (or 'ghost' gear) is a major threat to aquatic reptiles. Sea turtles in particular are at risk of becoming entangled in 'ghost' gear, such as nets, lines and ropes resulting in injury or death (Stelfox *et al.* 2016). Attempts may be made to remove derelict gear from reptiles found entangled to improve survival. This may require specialist techniques, tools and training. Injuries or wounds caused by entanglement may also require treatment.

For studies that look at the release of rehabilitated reptiles that were injured see *Species management – Rehabilitate and release injured or accidentally caught individuals*. For studies that look at the release of accidentally captured reptiles see *Threat: Biological resource use – Release accidentally caught ('bycatch') reptiles*.

Stelfox M., Hudgins J. & Sweet M. (2016) A review of ghost gear entanglement amongst marine mammals, reptiles and elasmobranchs. *Marine Pollution Bulletin*, 111, 6–17.

Sewage and wastewater

10.11. Improve treatment standards of sewage and wastewater

- We found no studies that evaluated the effects on reptile populations of improving treatment standards of sewage and wastewater.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

If left untreated, sewage and other wastewater may pose a threat to reptile populations, particularly those living in or around aquatic environments. Three main steps may be undertaken in the treatment of sewage and wastewater: primary treatment, where solid material that either floats or sinks is removed; secondary treatment, where soluble organic matter that escapes primary treatment is removed by microbial activity; and tertiary treatment, which provides a final stage to increase water quality before it is discharged into the environment. Improving the treatment standards for agricultural, residential and commercial waste may reduce the threat posed by wastewater to reptile populations.

10.12. Create walls or barriers to exclude pollutants

- We found no studies that evaluated the effects on reptile populations of creating walls or barriers to exclude pollutants.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Barriers may be used to prevent pollutants from entering waterways. Examples of such pollutants include garbage, sediments, oil, grease, excess nutrients, organic waste matter, pesticides and fertilisers. A range of options for both temporary and permanent structures have been suggested, including silt fences and gabions (Botting & Bellette 1998). Barriers that excluded pollutants could have positive effects on wild reptile populations.

Botting J. & Bellette K. (1998) *Stormwater pollution prevention: code of practice for local, state and federal government*. Environment Protection Authority, South Australia.

10.13. Cease or prohibit discharge of waste effluents overboard from vessels

- We found no studies that evaluated the effects on reptile populations of ceasing or prohibiting discharge of waste effluents overboard from vessels.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Discharge of waste effluents from vessels can contain pollutants. Reducing the amount of waste discharged from vessels may be beneficial for aquatic and semi-aquatic reptiles.

Oil spills

10.14. Establish emergency plans for oil spills

- We found no studies that evaluated the effects on reptile populations of establishing emergency plans for oil spills.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Oil spills are an obvious threat in marine and aquatic environments because the oil spreads quickly through the water column and can contaminate beaches and other coastal habitats. Oil spill emergency plans provide an overview of possible procedures, as well as details of which authorities to contact, should an oil spill occur. The aim of emergency plans is to increase the speed and effectiveness of the response to minimize harmful impacts (Li *et al.* 2016).

Li P., Cai Q., Lin W., Chen B. & Zhang B. (2016) Offshore oil spill response practices and emerging challenges. *Marine Pollution Bulletin,* 110, 6–27.

10.15. Contain or recover oil following spills

- We found no studies that evaluated the effects on reptile populations of containing or recovering oil following spills.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Oil spills are an obvious threat in marine and aquatic environments because the oil spreads quickly through the water column and can contaminate beaches and other coastal habitats. There are a number of different methods that can be used in an attempt to contain or recover the spill including: open-water mechanical recovery using booms and skimmers, offshore dispersant application using chemical dispersants, off-shore in situ burning, manual and mechanical clean-up to collect and dispose of contaminated sand or soil, and bioremediation using nutrients, aeration or bacteria to speed up natural breakdown of the oil (Huff & Shigenaka 2003). Different methods vary in number of ways, including cost, efficiency, time taken, amount of oil recovered, as well as impact on wildlife (Dave & Ghaly 2011).

Dave D. & Ghaly A.E. (2011) Remediation technologies for marine oil spills: a critical review and comparative analysis. *American Journal of Environmental Sciences*, 7, 423–440.

Huff R.Z. & Shigenaka G. (2003) Response considerations for sea turtles. Pages 49–68 in: G. Shigenaka (eds.) *Oil and Sea Turtles: Biology, Planning and Responses*. NOAA Ocean Service, Seattle, Washington.

10.16. Rehabilitate reptiles following oil spills

- **One study** evaluated the effects on reptile populations of rehabilitating reptiles following oil spills. This study was in the USA[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Survival (1 study):** One replicated study in the USA[1] found that almost all sea turtles that were de-oiled recovered and could be released.

BEHAVIOUR (0 STUDIES)

Background

Oil spills are an obvious threat in marine and aquatic environments because the oil spreads quickly through the water column. The oil clings to animals, remains long-term in the environment, and is often ingested. Heavily oil-covered or chronically oil-exposed turtles may have respiratory, skin and shell problems. Oiled turtles are also known to have increased white blood cell counts, reduced red blood cell counts, increased numbers of immature red blood cells, acute inflammation of skin and mucosal surfaces (Lutcavage *et al*. 1995). Long-term consequences of oil exposure are not well understood, but a high incidence of embryo deformity is known from turtle populations with chronic oil exposure (Bell *et al*. 2006). Other long-term indirect problems may include delayed mortality due to hindgut bacterial death in marine iguanas (Wikelski *et al*. 2002) and an increase in disease (Milton *et al*. 2003).

Reptiles may be rescued, rehabilitated and released back into the wild following oil spills to mitigate the negative effects of exposure.

For other studies relating to the re-release of injured reptiles or accidentally captured reptiles see *Species management – Rehabilitate and release injured or accidentally caught individuals* and *Biological resource use – Release accidentally caught ('bycatch') reptiles*.

Bell B., Spotila J.R. & Congdon J. (2006) High incidence of deformity in aquatic turtles in the John Heinz National Wildlife Refuge. *Environmental Pollution*, 142, 457–465.

Lutcavage M.E., Lutz P.L., Bossart G.D. & Hudson D.M. (1995) Physiologic and clinicopathologic effects of crude oil on loggerhead sea turtles. *Archives of Environmental Contamination and Toxicology*, 28, 417–422.

Milton S., Lutz P. & Shigenaka G. (2003) Oil toxicity and impacts on sea turtles. Pages 35–47 in G. Shigenaka (eds.) *Oil and Sea Turtles: Biology, Planning and Responses*. NOAA Ocean Service, Seattle, Washington.

Wikelski M., Wong V., Chevalier B., Rattenborg N. & Snell H.L. (2002) Galapagos Islands: marine iguanas die from trace oil pollution. *Nature*, 417, 607–608.

A replicated study in 2010 in two rehabilitation centres in Louisiana and Florida, USA (1) found that almost all sea turtles that received de-oiling treatment following the BP *Deepwater Horizon* oil spill in the Gulf of Mexico were rehabilitated and released back into the wild. Following de-oiling treatment, almost all rehabilitated sea turtles recovered and were released, including 189 of 192 Kemp's ridley *Lepidochelys kempii*, 112 of 113 green turtles *Chelonia mydas*, nine of nine loggerhead turtles *Caretta caretta* and five of five hawksbill turtles *Eretmochelys imbricata*). Three Kemp's ridley turtles died within 3 days and one green turtle was euthanised 142 days after admission (due to bacterial infection). Turtles (mainly juveniles with carapace length <25cm) were collected by crews patrolling the northern Gulf of Mexico and transported by vehicle from ports to rehabilitation facilities (1–3-hour journeys). Upon admission, turtles were weighed and measured (including blood samples). Turtles were de-oiled using multiple external cleanings using vegetable oil, mayonnaise and mild liquid detergent as well as oral doses of cod liver oil and oil. They were also provided with fluid therapy, and where necessary with vitamin B, iron and/or calcium supplements, antibiotics and veterinary treatment. A small number (15–20 individuals) also received oral charcoal. The dose and duration of petroleum exposure was unknown, but 139 turtles were classified as lightly oiled, 76 as moderately oiled, 46 as heavily oiled and 58 as severely oiled.

(1) Stacy N.I., Field C.L., Staggs L., MacLean R.A., Stacy B.A., Keene J., Cacela D., Pelton C., Cray C., Kelley M. & Holmes S. (2017) Clinicopathological findings in sea turtles assessed during the *Deepwater Horizon* oil spill response. *Endangered Species Research*, 33, 25–37.

10.17. Relocate reptiles (including eggs and hatchlings) following oil spills

- Studies investigating the effect of relocating reptiles are discussed in *Species management*.

Background

Relocating reptiles may be considered as a response of last resort following oil spills. Previous cases have involved moving thousands of sea turtle eggs or hatchlings (e.g. Safina 2011; Gaskill 2010), and as such great care should be taken when considering the potential risks and benefits that such large scale relocations may entail.

Gaskill M. (2010) *Turtle rescue plan succeeds*. Available at https://www.nature.com/news/2010/101008/full/news.2010.528.html?s=news_rss. Accessed 13 May 2021. doi:10.1038/news.2010.528

Safina C. (2011) The 2010 Gulf of Mexico oil well blowout: a little hindsight. *PLoS Biology,* 9: e1001049.

10.18. Regulate planning permission for gas/filling stations at reptile sites

- We found no studies that evaluated the effects on reptile populations of regulating planning permission for gas/filling stations at reptile sites.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Gas/filling stations present an important source for oil-based pollution to enter the surrounding environment (Hilpert *et al.* 2015). Regulating where gas/filling stations can be built may reduce the potential for pollutants to enter the environment in areas with important reptile habitats and may have positive effects on wild populations.

Hilpert M., Mora B.A., Ni J., Rule A.M. & Nachman K.E. (2015) Hydrocarbon release during fuel storage and transfer at gas stations: environmental and health effects. *Current Environmental Health Reports*, 2, 412–422.

Aquaculture effluents

10.19. Introduce and enforce water quality regulations for aquaculture systems

- We found no studies that evaluated the effects on reptile populations of introducing and enforcing water quality regulations for aquaculture systems.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Aquaculture systems may discharge water with waste and effluents into aquatic environments causing pollution and habitat degradation with adverse impacts on reptiles. Typical wastes include faeces, excess feed and nutrients, and chemicals, such as disinfectants, antifoulants, pesticides, herbicides, and drugs for disease control. Current water quality regulations at aquaculture systems vary widely between different countries, and some have no or very few regulations in place. Introducing and enforcing water quality regulations for aquaculture systems may reduce pollution and harmful impacts on reptile populations.

10.20. Switch to land-based aquaculture systems

- We found no studies that evaluated the effects on reptile populations of switching to land-based aquaculture systems.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Aquaculture systems may discharge water with waste and effluents into aquatic environments causing pollution and habitat degradation with adverse impacts on reptiles. Typical wastes include faeces, excess feed and nutrients, and chemicals, such as disinfectants, antifoulants, pesticides, herbicides, and drugs for disease control. Switching to land-based systems may lead to reductions in pollution, with positive impacts on reptile populations.

Agricultural and forestry effluents

10.21. Reduce pesticide, herbicide or fertilizer use

- We found no studies that evaluated the effects on reptile populations of reducing pesticide, herbicide or fertilizer use.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Agricultural land often receives high chemical inputs to control pests, weeds and fungal infections, and to fertilize crops. These chemicals also enter water bodies through spray drift or run off. These chemicals may impact reptiles through reducing plant and insect diversity, direct toxicity, and endocrine disruption.

The toxic effects of many organochlorine pesticides are well-studied in reptiles (e.g. DDT, diazion). Turtle species may be most risk from pesticide use because of their long life-span leading to bioaccumulation effects (Wagner *et al.* 2015). Endocrine disruptors in the environment are usually associated with polychlorinated biphenyl (PCB) and dichlorodiphenyltrichloroethane (DDT) contamination, both of which have been banned in many parts of the world. There are strong associations between organochlorine contamination and feminisation in alligators *Alligator mississippiensis* (Guillette *et al*. 1994) and mortality in turtles (Eisenreich *et al*. 2009). Despite the bans, problems for wildlife have continued due to the persistence of the contaminants in the environment.

The direct impact of fertilizers on reptiles may very between species (e.g. Marco *et al.* 2004; De Solla & Martin 2007), but by altering plant communities in waterways and other surrounding habitats, they may have important indirect impacts on reptile populations.

Organic farming, an agricultural system that excludes the use of synthetic fertilizers and pesticides and relies on techniques such as crop rotation, compost and biological pest control, is included within this intervention.

De Solla S.R. & Martin P.A. (2007) Toxicity of nitrogenous fertilizers to eggs of snapping turtles (*Chelydra serpentina*) in field and laboratory exposures. *Environmental Toxicology and Chemistry: An International Journal*, 26, 1890–1895.

Eisenreich K.M., Kelly S.M. & Rowe C.L. (2009) Latent mortality of juvenile snapping turtles from the upper Hudson River, New York, exposed maternally and via the diet to polychlorinated biphenyls (PCBs). *Environmental Science & Technology,* 43, 6052–6057.

Guillette L.J., Gross T.S., Masson G.R., Matter J.M., Percival H.F. & Woodward A.R. (1994) Developmental abnormalities of the gonad and abnormal sex hormone concentrations in juvenile alligators from contaminated and control lakes in Florida. *Environmental Health Perspectives*, 102, 680–688.

Marco A., Hidalgo-Vila J. & Díaz-Paniagua C. (2004) Toxic effects of ammonium nitrate fertilizer on flexible-shelled lizard eggs. *Bulletin of environmental contamination and toxicology*, 73, 125–131.

Pauli B.D., Money S. & Sparling D.W. (2010) Ecotoxicology of pesticides in reptiles. Pages 203–24 in: D.W. Sparling, G. Linder, C.A. Bishop & S. Krest (eds.) *Ecotoxicology of Amphibians and Reptiles, Second Edition.* CRC Press, Florida.

Randall N.P. & James K.L. (2012) The effectiveness of integrated farm management, organic farming and agri-environment schemes for conserving biodiversity in temperate Europe—a systematic map. *Environmental Evidence*, 1, 4.

Wagner N., Mingo V., Schulte U. & Lötters S. (2015) Risk evaluation of pesticide use to protected European reptile species. *Biological Conservation*, 191, 667–673.

10.22. Plant riparian buffer strips

- **One study** evaluated the effects of planting riparian buffer strips on reptile populations. The study was in the USA[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One replicated study in the USA[1] found that grassed riparian buffer strips were used by up to five snake species.

Background

Riparian buffer strips are uncultivated strips of permanent vegetation at the edge of waterways. They may be used within agricultural, forestry and urban systems to help reduce bank erosion and prevent pollutants and sediment entering the waterway. These buffer strips can therefore help to protect aquatic and semi-aquatic species.

For other studies that investigated retaining riparian buffers see *Threat: Biological resource use – Logging and wood harvesting – Retain riparian buffer strips during timber harvest* and *Habitat protection – Retain buffer zones around core habitat*.

A replicated study in 2003 of waterways in crop fields in Iowa, USA (1) found that over half of grassed riparian buffer strips were used by snakes. In total, 24 of 31 grassed riparian buffer strips were used by up to five snake species. Brown snakes *Storeria dekayi* (2.9 snakes/100 coverboards) and eastern garter snakes *Thamnophis sirtalis* (1.8 snakes/100 coverboards) were most abundant, followed by plains garter snake *Thamnophis radix* (0.8 snakes/100 coverboards), smooth green snakes *Lioclonorophis vernalis* (0.5 snakes/100 coverboards) and fox snakes *Elaphe vulpine* (0.4 snakes/100 coverboards). In May–August 2003, snakes were surveyed in 31 grassed waterways (>400 m apart) in crop fields (corn *Zea mays* and soybean *Glycine max*) that were established as part of the US government's National Conservation Buffer Initiative. Surveys were carried out using wooden coverboards (0.9 x 0.9 m) placed along each waterway (4–5 coverboards/waterway, 150 total coverboards). Observers checked each waterway for snakes weekly (12 checks/waterway, 1,800 total coverboard surveys).

(1) Knoot T.G. & Best L.B. (2011) A multiscale approach to understanding snake use of conservation buffer strips in an agricultural landscape. *Herpetological Conservation and Biology*, 6, 191–201.

10.23. Establish aquaculture facilities to extract the nutrients from agricultural run-off

- We found no studies that evaluated the effects on reptile populations of establishing aquaculture facilities to extract the nutrients from agricultural run-off.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Intensive agriculture constitutes a major source of pollution to marine and freshwater environments. Agricultural waste and pollutants can enter rivers and other watercourses and be discharged into the sea. Excess nutrients from agricultural waste can lead to diminished water quality and eutrophication events, including harmful algal blooms. Some species used in aquaculture can naturally improve water quality through feeding (e.g. filter feeding species, such as mussels) or through photosynthesis (e.g. algae species). Establishing certain types of aquaculture near polluted areas may help to remove excess nutrients (Duarte & Krause-Jensen 2018).

Duarte C.M. & Krause-Jensen D. (2018) Intervention options to accelerate ecosystem recovery from coastal eutrophication. *Frontiers in Marine Science,* 5, 470.

10.24. Treat wastewater from intensive livestock holdings

- We found no studies that evaluated the effects on reptile populations of treating wastewater from intensive livestock holdings.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Intensive agriculture constitutes a major source of pollution, particularly to aquatic environments. Wastewater from intensive livestock holdings containing bacteria, excess nutrients, chemical residues, and solid particles can enter rivers and other watercourses and be discharged into the sea. Treating wastewater from intensive livestock holdings may reduce the pollution levels in aquatic environments, and therefore reduce the associated impacts on aquatic reptiles.

Industrial pollution

10.25. Augment ponds with ground water to reduce acidification

- We found no studies that evaluated the effects on reptile populations of augmenting ponds with ground water to reduce acidification.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background
Disturbance of soils during land clearing or development can result in release of salts and therefore water with a low pH. This acidic water can end up in water bodies and have significant effects on aquatic biodiversity including reptiles. Adding uncontaminated ground water to ponds can help to regulate the pH of the water.

Studies that investigated regulating water levels of ponds are discussed in *Threat: Natural system modifications – Regulate water levels*.

10.26. Cease or prohibit the disposal of mining waste (tailings) at sea or in rivers

- We found no studies that evaluated the effects on reptile populations of ceasing or prohibiting the disposal of mining waste (tailings) at sea or in rivers.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background
Mine tailings, which are the ore waste of mines, typically in the form of a mud-like material, originate from both coastal and land-based mining activities and can be disposed of in aquatic environments causing chemical contamination. Ceasing or prohibiting the disposal of mining waste at sea or in rivers may reduce pollution and potential harm to aquatic and semi-aquatic reptiles.

10.27. Cease or prohibit the disposal of drill cuttings at sea or in rivers

- We found no studies that evaluated the effects on reptile populations of ceasing or prohibiting the disposal of drill cuttings at sea or in rivers.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Drill cuttings from oil and gas drilling activities are often discharged onto the sea floor or riverbed to form a cuttings pile. Drill cuttings consist of fragments of rock contaminated with drilling fluids, oil, and chemicals, which may have adverse impacts on aquatic and semi-aquatic reptiles. Ceasing or prohibiting the disposal of drill cuttings at sea or in rivers may reduce pollution and potential harm to reptiles.

10.28. Remove coal combustion waste to reduce contamination of terrestrial and aquatic habitats

- We found no studies that evaluated the effects on reptile populations of removing coal combustion waste to reduce contamination of terrestrial and aquatic habitats.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Fly-ash (flue ash, pulverised fuel ash) is a by-product of coal combustion and is often stored on or near coal fired power stations either in water impoundments (ponds) or underground in landfill. Fly-ash contains a number of metals and other trace elements and is known to contaminate groundwater. Metals and trace elements from fly-ash may be accumulated in the bodies of a range of reptile species with potentially harmful effects (Rowe *et al.* 2002). Removing the waste from coal combustion may help to reduce the exposure of reptiles to these harmful pollutants.

Rowe C.L., Hopkins W.A. & Congdon J.D. (2002) Ecotoxicological implications of aquatic disposal of coal combustion residues in the United States: a review. *Environmental monitoring and assessment*, 80, 207–276.

10.29. Set regulatory ban on marine burial of persistent environmental pollutants, including nuclear waste

- We found no studies that evaluated the effects on reptile populations of setting regulatory bans on marine burial of persistent environmental pollutants, including nuclear waste.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Historically, a range of persistent environmental pollutants have been intentionally disposed of in the marine environment. The disposal of nuclear and radioactive waste at sea was practised by 13 countries from 1946 until 1993, and other pollutants, including the pesticide dichlorodiphenyltrichloroethane (DDT), have been disposed of at sea in large quantities. A range of national laws and international treatise have sought to end the disposal of such pollutants into the marine environment, though enforcement is lacking in parts of the world, where illegal dumping is reported to occur. Setting pre-emptive regulatory bans on the sub-sea burial of persistent environmental pollutants may help prevent the occurrence of associated threats to aquatic and semi-aquatic reptiles.

Light pollution

10.30. Regulate artificial lighting during vulnerable periods

- We found no studies that evaluated the effects on reptile populations of regulating artificial lighting during vulnerable periods.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Artificial lighting may have a negative impact on a range of reptile species (Perry & Fisher 2006). Seasonal reduction of lighting to coincide with vulnerable periods, such as turtle breeding times, has been suggested as a conservation intervention, although, in practice, public safety considerations may often preclude action (Bertolotti & Salmon, 2015).

See also: *Enforce compliance to lighting regulations.*

Bertolotti L. & Salmon M. (2005) Do embedded roadway lights protect sea turtles? *Environmental Management*, 36, 702–10.

Perry G. & Fisher R.N. (2006) Night lights and reptiles: observed and potential effects. Pages 169–191 in: C. Rich & T. Longcore (eds.) *Ecological consequences of artificial night lighting*. Island Press, Washington D.C.

10.31. Enforce compliance to lighting regulations

- We found no studies that evaluated the effects on reptile populations of enforcing compliance to lighting regulations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

While the effects of light pollution on reptiles such as sea turtles are well-studied and many countries have policies prohibiting the lighting of beaches during the sea turtle nesting season. Enforcement of regulations may be required where compliance is low. This may involve surveillance, policing, and prosecution of offenders.

See also: *Regulate artificial lighting during vulnerable periods*.

10.32. Avoid illuminating key habitats

- We found no studies that evaluated the effects on reptile populations of avoiding illuminating key habitats.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background
Key habitats such as nesting and foraging sites should be left unlit to avoid disturbance to reptiles. Dark buffer zones may also be retained around them.

10.33. Use barriers or vegetation to reduce artificial light

- **One study** evaluated the effects of using barriers or vegetation to reduce artificial light on reptile populations. This study was in India[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (1 STUDY)

- **Behaviour change (1 study):** One replicated, site comparison study in India[1] found that when casuarina plantations were in close proximity (50 m) to the high tide line, more olive ridley turtle hatchlings crawled towards the sea compared to when plantations were 500 m from the high tide line, or when there was high intensity light and no barrier.

Background
Artificial light disrupts the natural sea finding behaviour of sea turtle hatchlings as hatchlings orient towards the lowest horizon with the brightest lights. As a result, high sea walls or trees can be used to block artificial inland lighting and in doing so mitigate the impact of artificial lighting on sea turtle hatchlings (Limpus & Kamrowski 2013).

Limpus C. & Kamrowski R.L. (2013) Ocean-finding in marine turtles: the importance of low horizon elevation as an orientation cue. *Behaviour*, 150, 863–93.

A replicated, site comparison study (years not provided) on a sandy beach in Orissa, India (1) found that when casuarina *Casaurina equisetifolia* plantations were in close proximity to the high tide line, more olive ridley *Lepidochelys olivacea* sea turtle hatchlings oriented themselves towards the sea compared to when plantations were further away from the tide line, or there was no light barrier. Fewer hatchlings oriented towards land and showed significant seaward orientation when casuarina were planted 50 m from the high tideline (0 of 10 hatchlings/trial oriented landwards) compared to when plantations were 500 m from the high tideline (4 of 10 hatchlings/trial) or where there was no light barrier (high intensity artificial lights visible: 5 of 10 hatchlings/trial; spaced out artificial lights visible: 2 of 10 hatchlings/trial). The 5 km beach was divided into areas with illumination and casuarina planted 50 m, or 500 m from the high tide line; no light barrier and lighting from well-spaced light from a highway; and no light barrier and high intensity artificial light. During the night, newly emerged hatchlings were placed in the middle of a 1.5 m circular arena with artificial light sources and the seaward horizon visible (nine trials/area, 10 hatchlings/trial). Hatchlings were allowed to orient, move to the edge of the arena and their direction of travel was recorded.

(1) Karnad D., Isvaran K., Kar C.S. & Shanker K. (2009) Lighting the way: towards reducing misorientation of olive ridley hatchlings due to artificial lighting at Rushikulya, India. *Biological Conservation*, 142, 2083–2088.

10.34. Use low intensity lighting

- **Four studies** evaluated the effects of using low intensity lighting on reptile populations. Three studies were in the USA[1-3] and one was in Malaysia[4].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (4 STUDIES)

- **Behaviour change (4 studies):** One replicated, controlled study in the USA[1] found that reducing the intensity of light sources did not improve loggerhead turtle hatchling seaward orientation. One replicated, site comparison study in Malaysia[4] found that green turtle hatchlings in low and moderate ambient artificial light took more direct crawl routes to the sea than hatchlings released in high ambient artificial light. One replicated, controlled study in the USA[3] found that in laboratory trials, loggerhead and green turtle hatchlings showed reduced preference for lower intensity light sources. One replicated, site comparison study in the USA[2] found mixed effects of embedding streetlights in the road on seaward orientation of loggerhead turtle hatchlings compared to overhead lighting depending on shading by shrubs and weather and lunar phase.

Background

Sea turtle hatchlings generally orient away from high silhouettes (dunes, trees) and towards brighter, lower horizons (light reflecting on the ocean). Artificial light is generally brighter than natural light from the moon and stars and not scattered by the atmosphere and thus interferes with the sea-orienting behaviours of sea turtle hatchlings. Using lower intensity lighting may mitigate some of the impact of increased levels of ambient artificial light near to nesting beaches.

See also: *Change the colour (spectral composition) of lighting*.

A replicated, controlled study (years not provided) on a beach in Florida, USA (1) found that using low rather than high intensity lighting did not improve loggerhead turtle *Caretta caretta* hatchling orientation seawards in any of five types of commercially-available artificial light types, but that one of 10 lighting treatments showed similar ranges of orientation to no lighting. Only high intensity yellow tinted incandescent lighting did not affect hatchling orientation compared when no lighting was used (results presented as crawl angles, see paper for details). Only low intensity low-pressure sodium vapour light did not affect the variation in crawl angle (results presented as crawl angles, see paper for details). Hatchlings in trials with white light sources (low and high intensity) showed the worst seaward orientation (results presented as crawl angles, see paper for details). Five commercially available lights were trialled at low and high intensities: high-pressure sodium vapour, low-pressure sodium vapour, yellow-tinted incandescent, red-tinted incandescent and white quartz. Trials were conducted at night by releasing 30 hatchlings/trial (from 30 different clutches) into the centre of an 8 m diameter sand arena divided into 32 segments, with segment one closest to the sea (0°) and a light positioned 4 m from the eighth segment (90°). After five minutes, the segment location of hatchlings was recorded. Trials were also carried out with no lighting.

A replicated, site comparison study in 2001 on a sandy beach in Florida, USA (2) found that when street lighting was embedded in the road, loggerhead turtle *Caretta caretta* hatchling orientation was more often seawards compared to when overhead lighting was used, though results depended on whether there was a full moon or new moon. Hatchling orientation was similar between embedded and overhead lighting conditions (seawards) in both unshaded and shaded sites during full moons. During the new moon, hatchlings moved seawards at all three sites when embedded lights were used, whereas with overhead lighting, the direction of movement was seaward in shaded areas, but more mixed in unshaded areas. With embedded lights and cloud cover, fewer turtles oriented directly eastwards compared to when there was no cloud cover. The nesting beach was parallel to a road with streetlights. In July–September 2001, loggerhead turtle hatchling orientation was compared between embedded lighting (LED lights in road studs at 9 m intervals) and overhead lighting (150W high-pressure sodium vapour angled away from the beach, 7.5–9 m high on wooden poles at 60–100 m intervals) in two unshaded sites and one site shaded by shrubs. Newly emerged hatchlings were placed in a 4 m diameter arena and exit direction from the arena was recorded. Trials were carried out 2–3 times for each site, lighting condition and moon phase (24 hatchlings from two or more nests/trial, sourced from 76 total nests).

A replicated, controlled study in 2000–2001 in a laboratory in Florida, USA (3) found that lowering the intensity of a light source led to lower preference for moving towards that light source in laboratory trials with loggerhead *Caretta caretta* or green *Chelonia mydas* turtle hatchlings. Hatchlings preferentially crawled towards unfiltered light over orange or red light in four of four trials, but preference for unfiltered light decreased as its intensity decreased (from 2 of 4 trials to 0 of 4 trials with decreasing intensity). At the lowest intensity, turtles showed preference for the filtered light in two of four trials. In comparisons between orange and red filtered light, turtles showed no preference in six of eight trials, but preferred red light when the orange light intensity was reduced to its lowest level (2 of 2 trials). Hatchlings were presented with a choice of two lights to crawl towards. High pressure sodium vapour lights were covered with an orange, red or no filter and the less filtered light in each comparison was tested at four intensity levels. The highest intensity used matched that of a beach adjacent streetlight located 40 or 60 m from a nest, and intensity was reduced using a neutral density filter. In 2000–2001, hatchlings were obtained from two beaches, with 25 used in each trial. Each hatchling was used in only one trial and then released in to the wild.

A replicated, site comparison study in 2005 on a sandy beach on the east coast of Terengganu, Peninsular Malaysia (4) found that green turtle *Chelonia mydas* hatchlings from a hatchery released in low and moderate ambient artificial light took a more direct route to the sea than hatchlings released in high ambient artificial light. Green turtle hatchlings released in low and moderate artificial ambient light dispersed at an angle similar to a direct line towards to the sea (62–76° dispersal angle), while hatchlings released in high ambient light dispersed at a different angle to the most direct route (148° dispersal angle, see original paper for information about directionality). In July–October 2005, sea-finding behaviours were tested on 30 hatchlings from each of 14 hatchery nests (420 hatchlings) under three different ambient lighting scenarios–high ambient light (300 m north of hatchery), moderate ambient light (directly in front of hatchery) and low ambient light (500 m south of hatchery; 10 hatchlings/nest/lighting scenario). All trials were conducted in an 8m wide circular sand arena 20 m from the sea with hatchlings placed under a bucket in the centre before release. The angle of dispersal for each hatchling was calculated using a compass from the centre to the point where the hatchling exited the arena and compared to the angle of the direct route to the sea.

(1) Witherington B.E. & Bjorndal K.A. (1991) Influences of artificial lighting on the seaward orientation of hatchling loggerhead turtles *Caretta caretta*. *Biological Conservation*, 55, 139–149.
(2) Bertolotti L. & Salmon M. (2005) Do embedded roadway lights protect sea turtles? *Environmental Management*, 36, 702–10.
(3) Sella K.N., Salmon M. & Witherington B.E. (2006) Filtered streetlights attract hatchling marine turtles. *Chelonian Conservation and Biology*, 5, 255–261.
(4) van de Merwe J.P., Ibrahim K. & Whittier J.M. (2013) Post-emergence handling of green turtle hatchlings: improving hatchery management worldwide. *Animal Conservation*, 16, 316–23.

10.35. Change the colour (spectral composition) of lighting

- **Three studies** evaluated the effects of changing the colour (spectral composition) of lighting on reptile populations. Two studies were in the USA[1,2] and one was in Australia[3].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (3 STUDIES)

- **Behaviour change (3 studies):** Two replicated, controlled studies (including one randomized study) in the USA[1] and Australia[3] found that yellow-tinted incandescent lighting did not affect the seaward orientation of loggerhead turtle hatchlings, whereas four other types of lighting did[1], and that hatchlings were disoriented in fewer trials by red lighting than by amber lighting[3]. One replicated, controlled study in the USA[2] found in laboratory trials that filtering out high wavelengths did not prevent loggerhead or green turtles crawling towards light sources.

Background

Sea turtle hatchlings generally orient away from high silhouettes (dunes, trees) and towards brighter, lower horizons (light reflecting on the ocean). Artificial light is generally brighter than natural light from the moon and stars and not scattered by the atmosphere and so may interfere with the sea-orienting behaviours of sea turtle hatchlings. Changing the spectral composition of artificial light, for example using yellow rather than white lighting, may therefore mitigate some of the impact of increased levels of ambient light near to nesting beaches.

See also: *Use low intensity lighting*.

A replicated, controlled study (years not provided) on a beach in Florida, USA (1) found that yellow-tinted incandescent lighting did not affect loggerhead turtle *Caretta caretta* hatchling seaward orientation at high intensity, whereas four other artificial lights did at both low and high intensities. Hatchlings released under high, but not low, intensity yellow-tinted incandescent lighting oriented in a similar direction compared to no lighting (data reported as crawl angle), which was directly seawards on average. High-pressure sodium vapour, low-pressure sodium vapour, red-tinted incandescent and white quartz lighting all affected sea turtle hatchling seaward orientation at high and low intensities compared to no lighting (data reported as crawl angle, see paper for details). Overall, hatchlings tended to be most attracted to white quartz lighting. High-pressure sodium vapour, low-pressure sodium vapour, yellow-tinted incandescent, red-tinted incandescent and white quartz lights were trialled at high and low intensity. Trials were conducted at night by releasing 30 hatchlings/trial (from 30 different clutches) into the centre of an 8 m diameter sand arena divided into 32 segments, with segment one closest to the sea (0°) and a light positioned 4 m from the eighth segment (90°). After five minutes, the segment location of hatchlings was recorded. Trials were also carried out with no lighting.

A replicated, controlled study in 2000–2001 in a laboratory in Florida, USA (2) found that filtering lights did not prevent loggerhead *Caretta caretta* or green *Chelonia mydas* turtle hatchlings from crawling towards a light source, though fewer turtles crawled towards filtered compared to unfiltered lights. Turtles crawled preferentially towards an orange, red or unfiltered light source, but showed no directional preference when no light source was present (data reported as average crawl angle). In addition, more turtles crawled towards a filtered light source over no light in two of four trials (68–84% of individuals), and towards an unfiltered light source over no light in two of two trials (96–100% of individuals). In 2000, turtles were placed in a circular arena where they could crawl in any direction, and in 2001 in a "T-maze" where they could crawl in one of two directions. Light sources were presented at the edge of the arena (orange, red, unfiltered or no light; 30 turtles/treatment) or in one arm of the T-maze (orange, red or unfiltered light; 25 turtles/treatment). High pressure sodium vapour lights were used that mimicked streetlights adjacent to turtle nesting sites (equivalent to 40 or 60 m away). In 2000, hatchlings were obtained from a beach hatchery, and in 2001 from natural nests. Each hatchling was used in only one trial and then released in to the wild. Behaviours were monitored with a video camera and monitor.

A replicated, randomized, controlled study in 2013 on unlit beaches in Queensland, Australia (3) found that loggerhead turtle *Caretta caretta* hatchlings were disoriented by amber 'turtle-safe' artificial lights and only disoriented by red 'turtle-safe' lights when three torches were used. Fewer hatchlings oriented directly seawards under amber lighting (72–89%), compared to no lighting (97%; statistical significance depended on the number of torches used, see original paper). Under red lighting, overall the proportion of hatchlings that oriented seawards was similar (98%) compared to no lighting (99%), but when three torches were used hatchlings were significantly more likely to orient towards the artificial light (see paper for details). Amber (620 nm peak wavelength, 9.8 light intensity) and red (640 nm, 8.3) 'turtle-friendly' LED lights were tested (1–4 torches/trial) during different parts of the lunar cycle and results compared to no lighting (amber: 21 total trials, red: 9). Trials were carried out by releasing <1-day-old hatchlings (20 hatchlings/trial) in an 8 m circular sand arena divided into 12 segments (see paper for more details). Hatchling segment location was recorded after five minutes.

(1) Witherington B.E. & Bjorndal K.A. (1991) Influences of artificial lighting on the seaward orientation of hatchling loggerhead turtles *Caretta caretta*. *Biological Conservation*, 55, 139–149.
(2) Sella K.N., Salmon M. & Witherington B.E. (2006) Filtered streetlights attract hatchling marine turtles. *Chelonian Conservation and Biology*, 5, 255–261.
(3) Robertson K., Booth D.T. & Limpus C.J. (2016) An assessment of 'turtle-friendly' lights on the sea finding behaviour of loggerhead turtle hatchlings. *Wildlife Research*, 43, 27–37.

Noise pollution

10.36. Impose noise limits in proximity to reptile habitats and routes

- We found no studies that evaluated the effects on reptile populations of imposing noise limits in proximity to reptile habitats and routes

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Noise pollution has been identified as having negative effects on many species of wildlife (Francis & Barber 2013). Excessive noise may have adverse impacts on reptile populations, with the potential to cause both avoidance behaviours and physical damage to individuals (Nelms *et al*. 2016). Imposing noise limits may reduce the potential for harm caused to wild reptiles.

Francis C.D. & Barber J.R. (2013) A framework for understanding noise impacts on wildlife: an urgent conservation priority. *Frontiers in Ecology and the Environment*, 11, 305–313.
Nelms S.E., Piniak W.E., Weir C.R. & Godley B.J. (2016) Seismic surveys and marine turtles: An underestimated global threat? *Biological Conservation*, 193, 49–65.

10.37. Install sound barriers in proximity to reptile habitats

- We found no studies that evaluated the effects on reptile populations of installing sound barriers in proximity to reptile habitats.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background
Sound barriers such as fences, walls or embankments may be installed in proximity to reptile habitats to reduce noise levels. Specially designed barriers that reflect or absorb sound are available. A buffer of trees and vegetation may also be used.

11. Climate change and severe weather

Background
Climate change and severe weather events are long-term and large-scale threats. As reptiles often tolerate relatively narrow ambient temperature ranges and have limited ability to disperse to alternative suitable habitats, global climate change may make some reptile populations vulnerable (Araújo *et al*. 2006). Changes to ambient temperatures may also affect the survival, performance and sex-ratio of reptile hatchlings in the wild (Fisher *et al*. 2014, Noble *et al*. 2018).

Many interventions used in response to climate change are general conservation interventions such as creating additional habitat to provide shade or refugia from extreme weather, maintaining ponds and wetlands to prevent desiccation, restoring eroded beaches, captive breeding and translocation. These interventions are discussed in: *Threat: Natural system modifications*, *Habitat protection*, *Habitat restoration and creation* and *Species management*.

Araújo M.B., Thuiller W. & Pearson R.G. (2006) Climate warming and the decline of amphibians and reptiles in Europe. *Journal of Biogeography,* 33, 1712–1728.

Fisher L.R., Godfrey M.H. & Owens D.W. (2014) Incubation temperature effects on hatchling performance in the loggerhead sea turtle (*Caretta caretta*). *PLoS One*, 9, e114880.

Noble D.W.A., Stenhouse V. & Schwanz L.E. (2018) Developmental temperatures and phenotypic plasticity in reptiles: a systematic review and meta-analysis. *Biological Reviews*, 93, 72–97.

11.1. Provide artificial shade for individuals

- **Two studies** evaluated the effects of providing artificial shade for individuals on reptile populations. One study was in Australia[1] and one was in Canada[2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (2 STUDIES)

- **Use (2 studies):** One replicated, randomized study in Australia[1] found that shaded, artificial rocky outcrops were used less often than unshaded ones by velvet geckos. One study in Canada[2] found that coverboards were used by northern pacific rattlesnakes in the year they were installed, but not a decade later.

Background
Providing artificial shade may mitigate effects of temperature increases by providing refugia for individuals in areas with extreme temperatures.

A replicated, randomized study in 1994–1995 on a sand plateau in New South Wales, Australia (1) found that reptiles tended to be found less often under artificial rocks (concrete pavers/paving stones) that were artificially shaded with cloth than under unshaded artificial rocks. Velvet geckos *Oedura lesueurii* used shaded pavers less frequently (9 pavers used by11 individuals) than unshaded pavers (28 pavers used by 45 individuals). One skink *Cryptoblepharus virgatus* and one broad-headed snake *Hoplocephalus bungaroides* were recorded in one unshaded paver each, but were not found in unshaded pavers. In November 1994–January 1995, artificial rocks (square concrete pavers: 19 cm wide, 5 cm thick) were placed in groups of four (20 cm apart in a square formation) at three undisturbed rock outcrops (sites >1km apart, 32–52 total pavers/site). Pavers were shaded or unshaded (90 x 50 cm steel frame covered with two layers of shade cloth; unshaded pavers had only steel frames), and were modified with either 4 mm or 8 mm crevices (created by gluing wood to the underside of the pavers). Surveys were attempted six times/site in April–November 1995 (18 total surveys) with reptiles marked with a toe clip. Human disturbance of artificial rocks prevented seven of 18 surveys from being carried out.

A study in 2007–2017 in shrub-steppe desert in the Okanagan Valley, Canada (2) found that coverboards provided as artificial shade after the installation of an exclusion fence were used by northern pacific rattlesnakes *Crotaus oreganus oreganus* in the year after the fence was installed, but there was no evidence that they were used 10 years later. Coverboards to provide shade during high temperatures were used by nine northern pacific rattlesnakes in the year they were installed. Nine to 10 years later, no snakes were found under the coverboards although 116 live snakes (northern pacific rattlesnake, great basin gophersnake *Pituophis catenifer deserticola*, and western yellow-bellied racer *Coluber constrictor mormon*) were captured along an adjacent exclusion fence over the same time period. In 2007, wooden coverboards (70 x 70 x 7 cm, 7 cm off the ground with 15–20 cm of sand excavated from underneath) were placed at 12 locations spaced at 30 m intervals along (360 m of) a 4 km long wire mesh snake exclusion fence (installed in 2006) to mitigate snake mortality due to heat exposure. At each interval, two coverboards were placed either side of the fence and one was placed 10–15 m away from the fence in natural habitat. Coverboard use was initially monitored in July 2007, and then monitoring was continued by mark-recapture surveys 5–6 days/week and walking the fence line 2–3 times/week in May–October 2016–2017.

(1) Webb J.K. & Shine R. (2000) Paving the way for habitat restoration: can artificial rocks restore degraded habitats of endangered reptiles? *Biological Conservation*, 92, 93–99.

(2) Eye D.M., Maida J.R., McKibbin O.M., Larsen K.W. & Bishop C.A. (2018) Snake mortality and cover board effectiveness along exclusion fencing in British Columbia, Canada. *Canadian Field-Naturalist*, 132, 30–35.

11.2. Provide artificial shade for nests or nesting sites

- **Four studies** evaluated the effects of providing artificial shade for nests or nesting sites on reptile populations. Two studies were in the USA[2,4] and one was in each of Panama[1], and Australia[3].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (3 STUDIES)

- **Reproductive success (3 studies):** One of two controlled studies (including one replicated study) in Panama[1] and Australia[3] found that shaded leatherback turtles nests had higher hatching success than unshaded nests. The other study[3] found that shaded and unshaded loggerhead turtle nests had similar hatching success. One replicated, controlled study in the USA[2] found that relocating diamondback terrapin nests to artificial nest mounds and providing shade had mixed effects on hatchling success.

- **Condition (2 studies):** One of two controlled studies (including one replicated study) in Panama[1] and Australia[3] found that greater shade cover resulted in smaller hatchlings for leatherback turtles. The other study[3] found that shading loggerhead turtle nests had mixed effects on hatchling size and crawl speed.

BEHAVIOUR (0 STUDIES)

OTHER (2 STUDIES)

- **Offspring sex ratio (2 studies):** One of two controlled studies (including one before-and-after study) in Panama[1] and the USA[4] found that shading leatherback turtle nests resulted in fewer female hatchlings compared to unshaded nests. The other study[4] found that shaded and unshaded Agassiz's desert tortoise nests produced a similar sex ratio of hatchlings.

Background

Providing shade may mitigate effects of temperature increases on incubating nests (e.g. Fuentes *et al*. 2012).

Studies investigating the effects of moving nests to improve hatching success, or artificially incubating eggs to manipulate sex ratios are discussed in *Species Management*.

Fuentes M., Fish M. & Maynard J. (2012) Management strategies to mitigate the impacts of climate change on sea turtle's terrestrial reproductive phase. *Mitigation and Adaptation Strategies for Global Change*, 17, 51–63.

A controlled study in 2007 in a beach hatchery on the south-western Caribbean Sea between Colombia and Panama (1) found that providing shading for leatherback turtle *Dermochelys coriacea* nests increased overall hatching success and reduced the number of female hatchlings compared to nests with no shading at all. Hatching success rates for shaded nests was higher (65–66% success) compared to unshaded nests (39%). Shaded nests produced fewer female hatchlings (40% shade: 25% females, 60% shade: 4% females; number of individuals not provided) than unshaded nests (100% female). Hatchlings from nests incubated in 60% shade were smaller than hatchlings from 40% shade or unshaded nests (see original paper for details). In total 36 nests were moved to a beach hatchery (15 x 8 m). Nests were incubated under one of three different levels of shade: 40% shade, 60% shade and no shade at all (12 nests/shade level). Shade was provided by materials of two different thicknesses stretched 1.5 m above ground.

A replicated, controlled study in 2006–2007 on an island on salt marsh grasses in New Jersey, USA (2) found that providing shade for diamondback terrapin *Malaclemys terrapin* nests relocated to artificial nest mounds resulted in lower hatching success compared to unshaded nests in one of six comparisons. Hatching success in loam and dredge mounds was similar for shaded (loam: 2 of 5 & 3 of 6 successful nests, 11 & 63% eggs hatched; dredge: 0 of 5 & 4 of 6 nests, 0 & 42% eggs) and unshaded nests (loam: 5 of 5 & 6 of 6 nests, 55 & 85% eggs; dredge: 0 of 5 & 6 of 6 nests, 0 & 59% eggs). In sand mounds, shaded nests had lower hatching success than unshaded mounds in the first year (shaded: 0 of 5 nests, 0% eggs; unshaded: 3 of 5 nests, 31% eggs) but similar success in the second (shaded: 3 of 6 nests, 41% eggs; unshaded: 6 of 6 nests, 65% eggs). Three experimental plots

(2.25 m²) were filled with 45 cm deep soil: sand from a beach; loamy sand from a natural nesting area or dredge soil from a nearby dried channel. One half of each plot was shaded by cloth 15 cm above the soil with the other half in full sun. Each nest was covered by a wire-mesh predator exclusion cage. Clutches of eggs were relocated to treatment plots from areas with high human activity (2006: 5 nests/treatment; 2007: 6 nests/treatment). Nests were excavated after 60 days to assess hatching success.

A replicated, controlled study in 2009–2010 on a beach in Queensland, Australia (3) found that shading loggerhead turtle *Caretta caretta* nests lead to larger hatchlings that moved and self-righted faster compared to unshaded nests, but hatching success and hatchling weight remained similar. Shaded hatchling carapace sizes were larger (1,545–1,597 mm²), hatchlings crawled faster (5–6 cm/second) and self-righted faster (2–3 s) than unshaded hatchlings (size: 1,484 mm², speed: 4 cm/second, self-righting time: 3 s), although shaded hatchling weight (20–21 g) was similar to unshaded hatchlings (20 g). Hatching success was similar between shaded nests (80–84 %) and unshaded nests (83%). Hatchlings were monitored in sites with three levels of artificial shading (4 h direct sun/day: eight clutches; 1.5 h direct sun/day: seven clutches; 0.5 h direct sun/day: seven clutches, average temperature of shaded nests: 30–31°) or in unshaded sites (9.5 h direct sun/day, six clutches, average temperature 32°). Artificial shade was provided with a rectangular synthetic shade cloth allowing 70% solar radiation positioned parallel to the shore. Clutches of eggs were collected from individual nesting loggerhead turtles in December 2009, eggs counted, and nests relocated. On emergence, hatchlings were weighed, measured and tested for crawl speed and righting responses before being released.

A controlled, before-and-after study in 2006–2009 in desert scrubland in California, USA (4) found that shaded Agassiz's desert tortoise *Gopherus agassizii* nests in a hatchery produced similar hatchling sex ratios as unshaded nests. Sex ratios were similar in shaded (0.2 females:1 male) and unshaded nests (0.3 females:1 male). This was despite soil temperatures in shaded areas being on average 13°C cooler mid-morning compared to unshaded areas. The authors report that the hatchling sex ratio from the previous three years of unshaded nests was the opposite way around (2–36 females:1 male) and that the change in sex ratio between the first three years and fourth year of the study could be explained by differences in air temperatures (see original paper for details). In spring 2006–2009, gravid, wild female Agassiz's desert tortoises were placed in individual pens in one of four predator-proof fenced enclosures with artificial nest burrows in a hatchery. After eggs were laid, tortoises were returned to their capture location. In 2009, fourteen pens were partially covered with 4 m² pieces of black shading cloth suspended 0.5 m above likely nest burrows. A further 10 nest burrows were left unshaded. In 2006–2008, twenty-nine to 37 hatchlings were sexed/year. In 2009, forty-six hatchlings from shaded burrows and 36 hatchlings from unshaded burrows were sexed.

(1) Patino-Martinez J., Marco A., Quiñones L. & Hawkes L. (2012) A potential tool to mitigate the impacts of climate change to the Caribbean leatherback sea turtle. *Global Change Biology*, 18, 401 ‑ 411.

(2) Wnek J.P., Bien W.F. & Avery H.W. (2013) Artificial nesting habitats as a conservation strategy for turtle populations experiencing global change. *Integrative Zoology*, 8, 209–221.

(3) Wood A., Booth D.T. & Limpus C.J. (2014) Sun exposure, nest temperature and loggerhead turtle hatchlings: Implications for beach shading management strategies at sea turtle rookeries. *Journal of Experimental Marine Biology and Ecology*, 451, 105–114.
(4) Nagy K.A., Kuchling G., Hillard L.S. & Henen B.T. (2016) Weather and sex ratios of head-started Agassiz's desert tortoise *Gopherus agassizii* juveniles hatched in natural habitat enclosures. *Endangered Species Research*, 30, 145–155.

11.3. Protect habitat along elevational gradients

- We found no studies that evaluated the effects of protecting habitat along elevational gradients on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background
Global warming is prompting poleward and uphill shifts in species' ranges (e.g. Chen *et al.* 2011). Species reliant on particular habitats may suffer population declines if they are unable to move towards higher latitudes and if there is no suitable habitat available at higher altitudes. Protecting habitat along elevational gradients may allow reptiles to naturally expand their ranges with climatic changes.

For other studies focusing on the impacts of protecting habitat on reptiles, see *Habitat Protection*.

Chen I.C., Hill J.K., Ohlemüller R., Roy D.B. & Thomas C.D. (2011) Rapid range shifts of species associated with high levels of climate warming. *Science*, 333, 1024–1026.

11.4. Use irrigation systems

- **Two studies** evaluated the effects of using irrigation systems on reptile populations. Both studies were in the USA[1,2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Reproductive success (1 study):** One replicated, controlled study in the USA[1] found that hatching success of Agassiz's desert tortoises was similar in irrigated and non-irrigated enclosures.
- **Survival (1 study):** One replicated, controlled study in the USA[1] found that survival of juvenile Agassiz's desert tortoises was similar in irrigated and non-irrigated enclosures.
- **Condition (2 studies):** Two replicated, controlled studies (including one paired study) in the USA[1,2] found that irrigating nests had mixed effects on growth of Agassiz's desert tortoises[1] and loggerhead turtles[2]. One of the studies[2] also found that loggerhead turtle hatchlings from nests that were irrigated were larger than those from non-irrigated nests.

BEHAVIOUR (0 STUDIES)

Background

Conservation of some species may require intensive management such as the redistribution of water resources. This could be achieved by using irrigation systems.

A replicated, controlled study in 2003–2008 in a desert scrub site in California, USA (1) found that irrigating the enclosures of Agassiz's desert tortoises *Gopherus agassizii* during a head-starting programme resulted in similar hatching success and juvenile survival but higher growth rate compared to tortoises with no supplementary water. Tortoises in irrigated and non-irrigated enclosures had similar hatching success (irrigated: 67% across 5 nests; non-irrigated: 77% across 7 nests) and juvenile survival (90% of 15 tortoises). Growth rate was higher in the first year in irrigated enclosures (15% increase/year) than in non-irrigated enclosures (4% increase/year). For tortoises hatched in 2003, those from irrigated enclosures were larger than those from non-irrigated enclosures after four years (irrigated: 81 cm; non-irrigated: 55 cm; 3 from each treatment; data from other years not provided). Enclosures were constructed in a natural habitat setting, of which six were irrigated (25–38 mm of water delivered through a sprinkler system three times in late winter-spring) and nine received only natural rain. Wild, adult females (number not given) were brought into the pens to lay eggs before being re-released. Hatching success was determined by counting hatchlings and un-hatched eggs, and hatchlings were marked, measured and re-measured after a year.

A replicated, controlled, paired study (year not given) on a sandy beach in Florida, USA (2) found that watering loggerhead turtle *Caretta caretta* nests lead to larger hatchlings that grew more in captivity over 10 weeks in one of three measures compared to hatchlings from un-watered nests. Hatchlings from watered nests had higher mass (watered: 17 g; un-watered: 16 g), straight carapace length (watered: 42 mm; un-watered: 40 mm) and straight carapace width (watered: 33 mm; un-watered: 32 mm) compared to those from un-watered nests. Growth in straight carapace width over 10 weeks was higher for hatchlings from watered compared to un-watered nests (watered: 65 mm; un-watered 62 mm after 10 weeks), though there was no significant difference in growth in mass (watered: 89 mm; not watered: 79 mm after 10 weeks) and straight carapace length (watered: 76 mm; not watered: 72 mm after 10 weeks). In one nesting season, eggs from 10 nests were divided in half and reburied 1 m apart (10 pairs of nests, buried 60 cm deep). One received 45 minutes of daily watering, while the other received no additional watering. Hatchlings (67 from watered, 55 from un-watered nests) were transferred to tanks supplied with fresh seawater, and measurements of mass, straight carapace length and width were taken weekly for 10 weeks.

(1) Nagy K.A., Hillard S., Dickson S. & Morafka D.J. (2015) Effects of artificial rain on survivorship, body condition, and growth of head-started desert tortoises (*Gopherus agassizii*) released to the open desert. *Herpetological Conservation and Biology,* 10 (Symposium), 535–549.

(2) Erb V., Lolavar A. & Wyneken J. (2018). The role of sand moisture in shaping loggerhead sea turtle (*Caretta caretta*) neonate growth in southeast Florida. *Chelonian Conservation and Biology*, 17, 245–251.

11.5. Reduce cumulative heating effects of urban development by planting vegetation

- We found no studies that evaluated the effects of reducing the cumulative heating effects of urban development by planting vegetation on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Cumulative effects of urban development (the 'urban heat island effect') may affect reptile physiology because of the narrow thermal tolerances of many species. Native vegetation plantings, green roofs and shade may allow for urban reptiles to persist in heat-polluted areas (Ackley *et al*. 2015).

Ackley J.W., Angilletta Jr M.J., DeNardo D., Sullivan B. & Wu J. (2015) Urban heat island mitigation strategies and lizard thermal ecology: landscaping can quadruple potential activity time in an arid city. *Urban Ecosystems,* 18, 1447–59.

12. Habitat protection

Background

Habitat destruction is the largest threat to biodiversity worldwide. Habitat fragmentation and degradation reduces both the amount and quality of remaining habitat. Therefore, habitat protection is one of the most necessary conservation interventions. Habitat, which in effect is the ideal place where resources exist for any given species, can be protected through the designation of legally protected areas using national or local area legislations. It can range from entire habitat protection (e.g. EU Habitats Directive 1992, USA Habitat Conservation Plans under the Endangered Species Act of 1973, and Environment Canada Protected Areas Strategy 2011) to community conservation with no formal protection or designation schemes. It can be difficult, if not impossible, to measure the effectiveness of legal protection on an area as there are usually not suitable comparisons. For example, monitoring generally only begins once the designation to a protected area comes into effect, meaning pre-designation data often do not exist, and the best quality habitats are often those selected for protection, meaning a similar unprotected habitat is not available as a comparison.

12.1. Protect habitat

Background

Legally protecting habitat may reduce its conversion and degradation by humans. This may in turn serve to maintain or slow the decline of the abundance and diversity of reptiles that make use of that habitat.

Assessing the effectiveness of protected areas is particularly difficult. For example, protected and unprotected areas used for comparison might start off with different quality habitats (protection often being granted to the best quality habitat). Protected areas are also more likely to be in remote areas, so less accessible to threats such as harvesting (Joppa & Pfaff 2009). Finally, effectiveness is best monitored over long timescales, but this increases the chance that other factors influence the ecosystem. The most reliable studies would compare similar quality protected and unprotected areas over time, and possibly correct for some of the biases.

Due to the number of studies found, this action has been split by species group.

Joppa L.N. & Pfaff A. (2009) High and far: biases in the location of protected areas. *PLoS ONE*, 4, e8273.

All reptiles (excluding sea turtles)

- **Seventeen studies** evaluated the effects of protecting habitat on reptile populations (excluding sea turtles). Four studies were in the USA[1,2,5,12], two were in each of Australia[6,9] and Brazil[10,17], and one was in each of Canada[3], Madagascar[4], South Africa[7], Spain[8], Hong Kong[11], Argentina[13], the borders of Zambia and Zimbabwe[14], Pakistan[15] and Mexico[16].

COMMUNITY RESPONSE (5 STUDIES)

- **Richness/diversity (5 studies):** Three of five studies (including two replicated, site comparison studies) in the USA[2], South Africa[7], Australia[9], Pakistan[15] and Mexico[16] found mixed effects of protected areas on reptile species richness[2,16] and combined reptile and amphibian species richness[15]. The other two studies[7,9] found that protected areas had higher reptile species richness than unprotected farmland.

POPULATION RESPONSE (16 STUDIES)

- **Abundance (13 studies):** Six of 11 studies (including five replicated, site comparison studies) in the USA[2,12], Canada[3], Hong Kong[11], Mexico[16], Australia[6,9], South Africa[7], Argentina[13], the border of Zambia and Zimbabwe[14] and Pakistan[15] found that protected areas had a higher abundance of reptiles[7,9], tortoises[12,13], Nile crocodiles[14] and combined reptiles and amphibians[15] than areas with less or no protection. Four studies found mixed effects of protection on the abundance of reptiles[2,3,16] and big-headed turtles[11]. The other study[6] found that water bodies in protected areas had fewer eastern long-necked turtles than those in suburban areas. One site comparison study in Brazil[10] found that areas with community-based management of fishing practices, which included protecting river turtle nesting beaches, had more river turtles than areas that did not manage fishing practices. One site comparison study in Madagascar[4] found that the abundance of different sized radiated tortoises in a protected area was more similar to that of an exploited population than to an unexploited population.
- **Occupancy/range (2 studies):** One replicated, site comparison study in Argentina[13] found that Argentine tortoises were found in one of two protected areas and two of three unprotected areas. One before-and-after study in Brazil[17] found that most reptile species were still present 20 years after an area was protected.
- **Survival (2 studies):** One replicated, randomized, site comparison study in the USA[12] found that in areas with greater protections, survival of Agassiz's desert tortoises was higher than in areas with less protections. One replicated, site comparison study in Spain[8] found that roads running through protected areas had more reptile road deaths than roads in unprotected areas.
- **Condition (4 studies):** Two of three site comparison studies (including one replicated study) in the USA[1], Australia[6] and Hong Kong[11] found that protected areas had larger red-eared sliders[1] and big-headed turtles[11] compared to areas where harvesting was allowed[1] or was thought to be occurring illegally[11]. The other study[6] found that eastern long-necked turtles in protected areas grew slower and were smaller than turtles in suburban areas. One site comparison study in Madagascar[4] found that radiated tortoises in a protected area had similar genetic diversity compared to populations outside of the protected area.

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One replicated study in the USA[5] found that a protected area was used by common chuckwallas.

A replicated, site comparison study in 1990–1991 in eight swamps and wetlands in southern Louisiana and western Mississippi, USA (1) found that sites protected from turtle harvesting and human disturbance had larger red-eared sliders *Trachemys scripta elegans* compared to harvested sites. Both male and female sliders in unharvested sites were larger (male: 19 cm carapace length;

female: 23) than in harvested sites (17, 18). While 24–28% of female turtles in unharvested sites were 24–28 cm long, there were no female turtles >22 cm long in harvested sites. In 1990–1991, turtles were captured from three protected (no public access or turtle harvesting), three public access (public access and no commercial harvesting) and two harvested sites (active commercial harvest). Turtles in protected and public access sites were trapped in baited hoop nets. Turtles in harvested sites were trapped by local hunters or purchased from local fish markets.

A replicated, site comparison study in 1994–1996 in desert shrub and grassland in south-central California, USA (2) found that lizard abundance and species richness was higher or similar inside a fenced protected area compared to outside depending on survey month and site. Lizard abundance was higher in three of six survey comparisons in a fenced protected area (4–10 lizards/transect) compared to outside of it (2–4 lizards/transect) but similar in the remaining three comparisons (inside: 2–5 lizards/transect; outside: 1–3 lizards/transect; see original paper for details). Lizard species richness was higher in one of six comparisons inside the protected area (2 species/transect) compared to outside of it (1 species/transect) but similar in the remaining five comparisons (inside: 2–3 species/transect; outside: 1–3 species/transect; see original paper for details). In 1994, two sites were selected near the north-eastern and southern boundary of the Desert Tortoise Research Natural Area (where off-road vehicles were prohibited from 1973, sheep grazing prohibited from 1978 and the boundary was fenced in 1980). Two 2.25 ha plots were established/site: one ≥400m inside the boundary and one outside the boundary (used by off-road vehicles until 1980 and grazed by sheep until 1994). In each plot, lizards were surveyed using 1.25 km transects in July 1994 and May and July 1995 (six surveys/site).

A replicated, before-and-after study in 1972–1973 and 2001–2002 in forest and wetlands in Ontario, Canada (3) found that decades of habitat protection had mixed effects on capture rates of native species and one introduced species was newly recorded. Results were not statistically tested. Thirty years after a first survey, capture rates tended to be higher for northern map *Graptemys geographica* (2002–2003: 0.004 individuals/hoop net night vs. 1972: 0.002) and stinkpot *Sternotherus odoratus* turtles (0.005 vs. 0.004). However, captures tended to be lower in 2002–2003 for painted *Chrysemys picta* (0.143 vs. 0.192), Blanding's *Emydoidea blandingii* (0.01 vs. 0.054), snapping turtles *Chelydra serpentina* (0.1 vs. 0.174) and spotted turtles *Clemmys guttata* (0 vs. 1 individual). Two spiny softshell *Apalone spinifera* and three introduced pond slider turtles *Trachemys scripta* were observed for the first time in 2002–2003. Turtle abundances were monitored in a 16 km^2 heavily trafficked National Park (gazetted in 1918, designated a Ramsar site in 1987, up to 500,000 visitors/year) in 1972–1973 and 2002–2003 using similar methods. Turtles were trapped using hoop (2002–2003: 3,237 hoop net trap nights; 1972: 522), basking and wire cage live traps as well as hand captures for two months in spring each year. In 2001–2002, all turtles were weighed, measured, sexed, and individually marked before release. Reported catch/trap nights above are for hoop net captures only.

A site comparison study in 2007 in dry tropical forest in southeast Madagascar (4) found that within a protected area, a population of radiated tortoises *Astrochelys radiata* had similar levels of genetic diversity as populations outside the protected area, but a size class distribution more similar to that of an exploited than unexploited population. Genetic diversity of the population within the protected area was similar to populations outside the protected area in 17 of 20 comparisons across four measures (data reported as diversity indices). The number of individuals of different size classes was similar in the protected area and exploited populations (>50% of population 0–4 kg in both), but different in the protected area compared to unexploited populations (>50% of population 4–8 kg or more in unexploited populations). In 2007, the protected area was searched by 5–7 people for a total of 10 days, and tortoises weighed, and a blood sample was taken. Genetic and size class data were compared with data from 12 other previously sampled populations, some that were exploited and some that were unexploited (numbers not provided).

A replicated study in 2009 in eight rock and shrub sites in a desert mountain preserve, Arizona, USA (5) found that a protected area surrounded by urban development contained signs of common chuckwalla *Sauromalus ater*. Common chuckwalla droppings were present in eight rock and shrub sites in a protected area (4–42 droppings/site). The authors report that dropping counts were correlated with plant diversity and the density of six plant species favoured by chuckwallas (see original paper for details). The eight sites in the protected area were rocky outcrops on ridges surrounded by urban development and had been protected for ~50 years prior to urban expansion. In the spring and autumn of 2009, faecal counts (used as an indicator of population size) at basking sites in a 1–2ha area within each reserve were obtained by a single observer over a 1 h period. Belt transects (1x10m) were used to assess crevice numbers, plant diversity and abundance.

A replicated, site comparison study in 2006–2007 in 15 wetlands in Australian Capital Territory, Australia (6) found that waterbodies in nature reserves had lower abundances of eastern long-necked turtles *Chelodina longicollis* compared to suburban areas and that adult turtles found in nature reserves grew slower and were more likely to be smaller than those in suburban areas. Eastern long-necked turtle abundance was 3 times lower in nature reserve waterbodies (15 turtles/wetland) compared to suburban waterbodies (44 turtles/wetland). Adult growth rates were more than five times lower in nature reserve waterbodies (0.2–0.3 mm/year) than in suburban waterbodies (1.3 mm/year). Smaller adult turtles (120–135 mm long) were found more often in nature reserves, whereas larger adult turtles (135–195 mm long) were found more often in suburban areas (see original paper for details). Turtles were monitored in seven waterbodies in nature reserves and eight in suburban areas within a 55 km^2 rural to urban gradient using baited crab traps. Traps were set in September and November 2006 and January and October 2007 (see original paper for details). Turtles were individually marked and measured prior to release.

A site comparison study in 2006–2007 in semiarid savanna with sparse woody vegetation in the southern Kalahari, South Africa (7) found that reptile species richness and abundance in protected areas was higher than in nearby unprotected farmland. Reptile species richness and abundance were higher in protected areas (richness: 3 species/transect, abundance: 6 individuals/transect) compared to unprotected farmland (richness: 1 species/transect, abundance: 2 individuals/transect). Ten of 11 reptile species were observed in the protected area compared to eight of 11 in unprotected farmland (see paper for details of individual species abundances). Reptiles were monitored in the Kgalagadi Transfrontier Park (38,000 km^2 formed from grasslands protected from 1931–1938) and 11 nearby commercial livestock farms (total unprotected area 10,000 km^2) in March–May 2006 and 2007. Reptile abundances and species richness was estimated along 500 x 5m transects (55 transects each in the protected and unprotected study areas) using visual encounter surveys with no movement of logs or leaf litter.

A replicated, site comparison study in 2002 in four regions of Catalonia, Spain (8) found that the roads in areas with high protection were more likely to have reptile roadkill than in areas with low or no protection. Results were presented as statistical model outputs (see original paper for details). In total, 127 reptiles were collected in spring and 118 reptiles were collected in autumn, the majority of which were Montpellier snakes *Malpolon monspessulanus*. Roads were surveyed for roadkill in 2002 fortnightly by car in spring (3 times between 14 April and 20 May) and autumn (3 times between 10 October and 15 November) along a randomly selected 20 km stretch of secondary road in each of 41 counties (246 surveys over 4,900 km). The protection status of the area around the road was categorised as high (Natural Park or National Park: 5 roads), low (areas of natural interest: 7 roads) or not protected (29 roads).

A replicated, site comparison study in 2007–2008 in two areas of mallee woodland in South Australia, Australia (9) found that reptile species richness and abundance was higher in conservation parks than in adjacent farmland. Reptile species richness and abundance were both higher within conservation parks (7 species/site; 18 individuals/site) than in adjacent farmland (4 species/site; 11 individuals/site), and on farmland, both richness and abundance declined with increasing distance from the conservation parks (results reported as statistical model outputs, see original paper for details and individual species abundances). In total, 431 reptiles of 31 species were counted. Reptiles were surveyed in mallee woodland (*Melaleuca uncinata* and *Eucalyptus* spp.) in two areas in the Eyre Peninsula in December 2007 and January–February 2008. Three replicated sampling blocks were surveyed/area and each block contained two sampling sites within the conservation park (50–750 m from the park boundary) and three sites in adjacent farmland (in remnant habitat in sand dunes or roadside verges, 7–12 km from the park boundary). Reptiles were sampled using 10 pitfall traps and drift fences spaced 25 m apart along a 225 m linear transect in each sampling site. Traps were open for six consecutive 24 hour periods/month.

A site comparison study in 2009 on a flood plain with lakes and channels in Pará, Brazil (10) found that areas with community-based management (CBM) of fishing practices, that included protecting turtle nesting beaches, limiting use of gill-nets, seasonal fishing restrictions, and a ban on turtle trading had more river turtles (*Podocnemis sextuberculata*, *Podocnemis unifilis* and *Podocnemis expansa*) than areas without CBM. The effect of different aspects of the management programme cannot be separated. Turtles were more abundant in areas with CBM (321 individuals) than in areas without CBM (33 individuals). For *Podocnemis sextuberculata*, abundance was higher in areas with CBM (14 individuals/1,000 m^2 netting/12 hours) than in areas without (2 individuals/1,000 m^2 netting/12 hours). Turtle biomass was also greater with CBM (20 kg/1,000 m^2 netting/12 hours) than without CBM (3 kg/1,000 m^2 netting/12 hours). The fishing agreement that formed the CBM programme had been in place for 20–30 years to protect nesting beaches and reduce illegal trade; though consumption of turtles was still permitted. While 13 communities in the area were a part of the fishing agreement, only two implemented the agreement. Turtle numbers were sampled at 14 sites (7 with CBM; 7 without CBM) in August–October 2009 using gill nets (15 nets/site; 215 m^2 nets; 3 each of 5 mesh sizes) with help from local fishers.

A site comparison study in 2009–2011 in freshwater lakes, rivers and streams in Hong Kong (11) found that big-headed turtles *Platysternon megacephalum* were larger in a stream inside a fenced, patrolled, protected area without turtle harvesting compared to turtles in four national park sites where illegal harvesting is believed to take place. Male and female big-headed turtles captured in the unharvested stream were larger (male: 123 mm long, female: 105 mm long) than male and female turtles in harvested streams (male: 91–104, female: 92–97). Male turtles in the unharvested site were significantly larger than females at the same site, whereas male turtles in harvested sites were of similar size to females in unharvested and harvested sites. In the unharvested site, male turtle density was higher (unharvested: 46 individuals/km; harvested: 3–35 turtles/km), female turtle density was similar (unharvested: 34; harvested: 3–35) and juvenile density was lower (unharvested: 38; harvested: 55–128) compared to harvested streams (results were not statistically tested). Turtles were surveyed in one unharvested stream in a fenced, patrolled conservation area and four streams in national parks where illegal harvesting was believed to take place. Between September 2009 and June 2011, visual encounter surveys (51 total surveys totalling 263 hours) and baited hoop trapping (10–20 traps/site, 5,124 total trapping hours) were carried out.

A replicated, randomized, site comparison study in 2011 in desert shrub and grassland in the western Mojave Desert, California, USA (12) found that in an area with the most human restrictions Agassiz's desert tortoises *Gopherus agassizii* were more abundant and had a lower mortality rate. Desert tortoise densities were approximately six-times higher in the most protected area, the Tortoise Natural Area (15 tortoises/km^2) than in designated tortoise critical habitat (2 tortoises/km^2) and four-times higher than on private lands (4 tortoises/km^2). Tortoise annual death rates over the preceding four years were estimated as lowest in the Tortoise Natural Area (3% mortality/year) compared to private lands (6%) and in critical habitat (20%, results were not statistically tested). Tortoises were surveyed in 240 1 ha plots across three different management areas (80 plots/area): Tortoise Natural Area (1973: closed to recreational vehicles; 1980: fully enclosed and closed to mining and livestock grazing, 2010: 12 km of fencing extended to prevent tortoises leaving), critical habitat areas (1994: recreational vehicle use restricted but not enforced with some annual closures, 1990: closed to sheep grazing) and private lands (unregulated sheep grazing, intensive recreational vehicle use, hunting and rubbish dumping). In April–May 2011 plots were surveyed on foot twice/day for live or dead tortoises and field signs.

A replicated, site comparison study in 2011–2012 in five sites of dry forest, brush and grassland or agricultural lands in La Pampa, Catamarca and Santiago del Estero, Argentina (13) found that Argentine tortoises *Chelonoidis chilensis* were present in one of two protected areas and two of three unprotected sites surveyed. Fourteen tortoises were present in one protected area and none in the second protected area. Three tortoises were counted in two unprotected sites surveyed (1 or 2 tortoises/site) and none in a third unprotected site. In November 2011–March 2012, Argentine tortoises were surveyed in two protected areas and three unprotected sites. Visual encounter surveys were carried out on foot using line transects (100 m long, 30 m wide) and covered 171,000 m^2 in each area or site (8–15 hours survey time/site).

A study in 2007 along a river on the borders of Zambia and Zimbabwe (14) found that abundance of Nile crocodiles *Crocodylus niloticus* was highest in river areas on the edge of national parks. Abundance of Nile crocodiles was higher in river reaches on the edge of national parks (21 crocodiles/km river) compared to areas with less protection (7 crocodiles/km river). In October 2007, Nile crocodiles were surveyed at night by boat using spotlights along a 262 km stretch of the Zambezi river. Two boats with three people (a navigator, recorder and observer) each surveyed a stretch at a time. The river edge bordered national parks (with 75 km^2/scout protection levels, use limited to ecotourism and mineral extraction), and other areas with less protection, including game management areas (with 122–123 km^2/scout protection levels and multiple legal natural resource uses) and open areas (>200 km^2/scout protection and uncontrolled natural resource use).

A replicated study in 2011–2013 in 22 sites of wetland, cropland, open scrub and forest in Punjab, Pakistan (15) found that protected sites overall tended to have higher numbers of reptiles and amphibians combined than unprotected sites, but that protected wetlands tended to have lower diversity of reptile and amphibian species combined compared to unprotected wetlands. Results were not statistically tested. Of 33 species that were observed overall, 19 were present in protected wetlands (total of 2,486 individuals), whereas 27 were present in unprotected wetlands (1,766 individuals). Nineteen species (6,586 individuals) were present in protected areas of mixed open habitat, whereas 15–21 species (154–2,092 individuals) were present in unprotected forest and croplands. Reptiles and amphibians were surveyed using visual encounter methods in 22 sites designated as: protected wetlands, unprotected wetlands, mixed open scrub and cultivated farmland protected as wildlife sanctuaries, and unprotected cropland and tropical thorny forest (number of sites/designation not provided). Surveys were carried out in each site in March–April, July–August, September–October and November–February from March 2011 to July 2013 (6 survey hours/day for 1–3 days at a time).

A site comparison study in 2012–2015 in semi-deciduous tropical forest and cultivated land in Nayarit, Mexico (16) found that inside a protected area the numbers of lizards counted was higher, but snakes and turtles lower than outside the protected area, and the number of species found was similar for lizards and turtles, but lower inside the protected area for snakes. Results were not statistically tested. Eleven lizard species were found both inside and outside of the protected area, and more individual lizards were counted inside (937 individuals) than outside (834 individuals) the protected area. Less snake species and less individual snakes were counted inside the protected area (species: 14; individuals: 30) compared to outside of it (species: 20; individuals: 64). Two turtle species were found both inside and outside of the protected area, but less individual turtles were counted inside (4 individuals) than outside (8 individuals) of the protected area. Reptiles were surveyed inside and outside of a natural protected area using visual encounter surveys on 39 occasions in June 2012–August 2015 (760 survey hours both inside and outside the park).

A before-and-after study in 2007–2015 in urban parkland with remnant forest in Pará state, Brazil (17) found that most lizard species and one of two amphisbaenian species recorded were still present 20 years after a park was protected. Twenty-two of 25 lizard species and one of two amphisbaenian species found in the park before 1985 were still present after 1990. Two lizard and two amphisbaenian species were recorded in the park after 1990 but not before 1985. A state park was protected from unsustainable resource use in 1993. The park (1,393 ha) included two lakes and remnant woodland and was an important recreational area for neighbouring urban areas. Reptiles were surveyed between March 2007 and January 2009 (81 total days of collecting and 48 days of pitfall trapping) and June 2014 and March 2015 (39 total days of collecting and pitfall trapping combined). Results were combined with herpetological collection records and historical survey data from 1990 onwards and compared with historical records and surveys undertaken before 1985.

(1) Close L.M. & Seigel R.A. (1997) Differences in body size among populations of red-eared sliders (*Trachemys scripta elegans*) subjected to different levels of harvesting. *Chelonian Conservation and Biology*, 2, 563–566.

(2) Brooks M. (1999) Effects of protective fencing on birds, lizards, and black-tailed hares in the western Mojave Desert. *Environmental Management*, 23, 387–400.

(3) Browne C.L. & Hecnar S.J. (2007) Species loss and shifting population structure of freshwater turtles despite habitat protection. *Biological Conservation*, 138, 421–429.

(4) Rioux Paquette S., Ferguson B.H., Lapointe F.J. & Louis Jr E.E. (2009) Conservation genetics of the radiated tortoise (*Astrochelys radiata*) population from Andohahela National Park, southeast Madagascar, with a discussion on the conservation of this declining species. *Chelonian Conservation and Biology*, 8, 84–93.

(5) Sullivan B.K. & Williams R.E. (2010) Common chuckwallas (*Sauromalus ater*) in urban preserves: do food plants or crevice retreats influence abundance. *Herpetological Conservation Biology*, 5, 102–110.

(6) Roe J.H., Rees M. & Georges A. (2011) Suburbs: Dangers or Drought Refugia for Freshwater Turtle Populations? *Journal of Wildlife Management*, 75, 1544–1552.

(7) Wasiolka B. & Blaum N. (2011) Comparing biodiversity between protected savanna and adjacent non-protected farmland in the southern Kalahari. *Journal of Arid Environments*, 75, 836–841.

(8) Garriga N., Santos X., Montori A., Richter-Boix A., Franch M. & Llorente G.A. (2012) Are protected areas truly protected? The impact of road traffic on vertebrate fauna. *Biodiversity and Conservation*, 21, 2761–2774.

(9) Williams J.R., Driscoll D.A. & Bull C.M. (2012) Roadside connectivity does not increase reptile abundance or richness in a fragmented mallee landscape. *Austral Ecology*, 37, 383–391.

(10) Miorando P.S., Rebêlo G.H., Pignati M.T. & Brito Pezzuti J.C. (2013) Effects of community-based management on Amazon river turtles: a case study of *Podocnemis sextuberculata* in the lower Amazon floodplain, Pará, Brazil. *Chelonian Conservation and Biology*, 12, 143–150.

(11) Sung Y.-H., Karraker N.E. & Hau B.C.H. (2013) Demographic evidence of illegal harvesting of an endangered Asian turtle. *Conservation Biology,* 27, 1421–1428.

(12) Berry K.H., Lyren L.M., Yee J.L. & Bailey T.Y. (2014) Protection benefits desert tortoise (*Gopherus agassizii*) abundance: the influence of three management strategies on a threatened species. *Herpetological Monographs*, 28, 66–92.

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(15) Rais M., Akram A., Ali S.M., Asadi M.A., Jahangir M., Jilani M.J. & Anwar M. (2015) Qualitative analysis of factors influencing the diversity and spatial distribution of herpetofauna in Chakwal Tehsil (Chakwal district), Punjab, Pakistan. *Herpetological Conservation and Biology*, 10, 801–810.
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(17) Avila-Pires T.C.S., Alves-Silva K.R., Barbosa L., Correa F.S., Cosenza J.F.A., Costa-Rodrigues A.P.V., Cronemberger A.A., Hoogmoed M.S., Lima-Filho G.R., Maciel A.O., Missassi A.F.R., Nascimento L.R.S., Nunes A.L.S., Oliveira L.S., Palheta G.S., Pereira Jr A.J.S., Pinheiro L., Santos-Costa M.C., Pinho S.R.C., Silva F.M., Silva M.B. & Sturaro M.J. (2018) Changes in amphibian and reptile diversity over time in Parque Estadual do Utinga, Para State, Brazil, a protected area surrounded by urbanization. *Herpetology Notes,* 11, 499–512.

Sea turtles

- **Four studies** evaluated the effects of protecting habitat on sea turtle populations. One study was in each of Costa Rica[1], the Seychelles[2], Belize[3] and the USA[4].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (4 STUDIES)

- **Abundance (3 studies):** One before-and-after study in Costa Rica[1] found that after an area was protected, there were fewer nesting female leatherback turtles than before protection. One replicated, randomized, site comparison study off the coast of Belize[3] found that in protected areas there were more hawksbill turtles than outside. One site comparison study in the USA[4] found that differences in the abundance of green, loggerhead and hawksbill turtles in protected and unprotected areas were mixed.
- **Reproductive success (2 studies):** One before-and-after study in Costa Rica[1] found that after an area was protected, more leatherback turtle hatchlings were produced than before protection. One before-and-after study in the Seychelles[2] found that nesting activity by green turtles increased following both habitat and species protection.

BEHAVIOUR (0 STUDIES)

A before-and-after study in 1988–2004 on three beaches in Costa Rica (1) found that six years after a national park was created, numbers of nesting female leatherback turtles *Dermochelys coriacea* tended to be lower and hatchling numbers tended to be higher than before the park was created. Results were not statistically tested. In the six nesting seasons after a national park was created, 68–1,000 female leatherback turtles nested/year and 15,734–153,547 hatchlings/year were produced, compared to 732–1,504 nesting female leatherback turtles and 30,180–30,788 hatchlings/year in the three years before the park's creation. The park was declared in 1991 and comprises three beaches. An unspecified number of nests were relocated due to threat of tidal inundation. Nesting female numbers were based on counting depressions left in the sand by nesting turtles.

A before-and-after study in 1968–1976 and 1981–2008 on sandy beaches on an atoll island, Aldabra Atoll, Seychelles (2) found that legal protection for green turtles *Chelonia mydas*, followed by protection of the whole island 15 years later, was associated with an increase in nesting activity. Results were not statistically tested, and the effects of species and habitat protection cannot be separated. Overall nesting activity was estimated to be higher 36–40 years after turtle protection began (2004–2008: 28,200 nesting attempts/year) compared to 13–17 years after turtle protection began (1981–1985: 10,900–16,500 nesting attempts/year). The authors also reported that estimates of nesting activity around the time that turtle protection began ranged from sightings of seven females (11 day survey in 1967), to 2,000–3,000 nests/year (surveys during 1968–1970 and 1975–1976). Protection for turtles began in 1968, with the Green Turtle Protection Regulations 1968, and the atoll became a UNESCO World Heritage Site in 1983. In 1981–2008, up to 68 nesting beaches on the atoll were surveyed for turtle tracks and evidence of nesting. Survey effort varied between different years and beaches, with beaches surveyed 0–37 times/years in 1981–1994, and 4–171 times/month in 1995–2008.

A replicated, randomized, site comparison study in 2009–2010 on an offshore coral reef atoll with two marine protected areas near Belize (3) found that hawksbill turtle *Eretmochelys imbricata* abundance was greater inside than outside protected areas. Hawksbill turtle abundance was greater inside protected areas (2–3 turtle sightings/hour) than outside protected areas (1 turtle sightings/hour). Hawksbill turtles were surveyed in the vicinity of a coral reef atoll (45 km long and 10 km wide) that contained six small cays and two no-take protected areas. Turtles were monitored on 49 randomly selected transects (1 km long) carried out over 30 days in April–May 2010 by three swimmers (1–20 m depths). In addition, 26 turtles were captured in April–May 2009 and in May 2010. Captured turtles were weighed and measured and a subset (10 individuals in 2009 and 9 individuals in 2010) were radio tracked every 24 hours for 6–25 days. It is unclear whether the captured turtles were included in the abundance estimates.

A site comparison study in 2003–2012 in shallow coastal and deeper water off the coast of Florida, USA (4) found that inside a protected area there were fewer green turtles *Chelonia mydas*, more loggerhead turtles *Caretta caretta* and similar numbers of hawksbill turtles *Eretmochelys imbricata* compared to outside of the protected area. Results were not statistically tested. Inside a protected area, 0.1–0.6 green turtles/km, 0.2–0.5 loggerhead turtles and 0.01–0.2 hawksbill turtles were encountered compared to 1.8 green turtles/km, 0.1 loggerhead turtles/km and 0.01 hawksbill turtles/km outside the protected area. Three sites (15–27 km^2) were surveyed in shallow-water habitats (0.2–6 m depths) inside a protected area (a national marine sanctuary covering 835 m^2 of open water and 8 km^2 on land) and compared to a single unprotected site (36 km^2) in deeper waters (3–6 m depths). Surveys were carried out during 27 boat trips in September 2003–September 2012 (139 total survey days) by driving haphazard, non-linear transects on a boat with several observers (129 km^2 total area covered by surveys). Turtle sightings were recorded and where possible turtles were captured, individually-marked, weighed and measured.

(1) Santidrián Tomillo P., Vélez E., Reina R.D., Piedra R., Paladino F.V. & Spotila J.R. (2007) Reassessment of the leatherback turtle (*Dermochelys coriacea*) nesting population at Parque Nacional Marino Las Baulas, Costa Rica: effects of conservation efforts. *Chelonian Conservation and Biology*, 6, 54–62.

(2) Mortimer J.A., Von Brandis R.G., Liljevik A., Chapman R. & Collie J. (2011) Fall and rise of nesting green turtles (*Chelonia mydas*) at Aldabra Atoll, Seychelles: positive response to four decades of protection (1968–2008). *Chelonian Conservation and Biology*, 10, 165–176.
(3) Scales K.L., Lewis J.A., Lewis J.P., Castellanos D., Godley B.J. & Graham R.T. (2011) Insights into habitat utilisation of the hawksbill turtle, *Eretmochelys imbricata* (Linnaeus, 1766), using acoustic telemetry. *Journal of Experimental Marine Biology and Ecology*, 407, 122–129.
(4) Herren R.M., Bagley D.A., Bresette M.J., Holloway-Adkin K.G., Clark D. & Witherington B.E. (2018) Sea turtle abundance and demographic measurements in a marine protected area in the Florida Keys, USA. *Herpetological Conservation and Biology,* 13, 224–239.

12.2. Retain connectivity between habitat patches

- **Two studies** evaluated the effects of retaining connectivity between habitat patches on reptile populations. One study was in Brazil[1] and one was in Madagascar[2].

COMMUNITY RESPONSE (2 STUDIES)

- **Community composition (2 studies):** One replicated, site comparison study in Brazil[1] found that forest fragments connected by corridors and isolated forest fragments had similar reptile species composition. One site comparison study in Madagascar[2] found that in an area with hedges connecting different habitat types, reptile communities were more similar across the different habitat types than in an area with no hedges.
- **Richness/diversity (2 studies):** One replicated, site comparison study in Brazil[1] found that forest fragments connected by corridors and isolated forest fragments had similar reptile species richness. One site comparison study in Madagascar[2] found that an area with hedges connecting different habitat types had more unique reptile species than an area without hedges.

POPULATION RESPONSE (1 STUDY)

- **Abundance (1 study):** One replicated, site comparison study in Brazil[1] found that forest fragments connected by corridors and isolated forest fragments had a similar abundance of reptiles, including leaf litter lizards.

BEHAVIOUR (0 STUDIES)

Background

Habitat fragmentation, as well as destruction, may be an important driver of population declines. Small areas hold fewer species than large ones and if individuals are unable to cross areas of converted habitat then populations in separate habitat patches will become isolated. This potentially makes them more vulnerable to extinction, from natural variations in birth and death rates or sex ratios, from inbreeding depression and from outside pressures, both natural (such as storms or wildfires) and man-made (such as hunting or continued habitat loss), although the precise effects of habitat fragmentation, as opposed to loss, are debated (e.g. Fahrig 1997).

Theoretically, the number of species surviving in a habitat fragment is determined by its size and its effective distance to other habitat patches (MacArthur & Wilson 1967). Maintaining habitat connectivity is therefore often seen as a way to increase the viability of populations, but there is considerable debate as to the effectiveness of such 'wildlife corridors' (e.g. Beier & Noss 1998).

Studies describing the effects of creating or restoring habitat, rather than retaining what is already there, are in *Habitat restoration and creation*.

Beier P. & Noss R.F. (1998) Do habitat corridors provide connectivity? *Conservation Biology*, 12, 1241–1252.

Fahrig L. (1997) Relative Effects of Habitat Loss and Fragmentation on Population Extinction. *The Journal of Wildlife Management*, 61, 603–610.

MacArthur R.H. (1967) *The Theory of Island Biogeography*. Princeton University Press, Princeton, N.J.

A replicated, site comparison study in 2002–2003 in forest in São Paulo state, Brazil (1) found that forest fragments connected with forest corridors did not have greater lizard species richness or abundance than isolated fragments. Reptile species composition, richness, total abundance and abundance of leaf litter lizards *Ecpleopus gaudichaudii* were similar between connected (richness: 1–4 species, total abundance: 10–15 individuals, total leaf litter lizard abundance: 3–15 individuals) and isolated forest fragments (richness: 2–3 species, total abundance: 11–33 individuals, total leaf litter lizard abundance: 2–29 individuals), regardless of fragment size (species composition results reported as model outputs). Forest fragments (2–48 ha) in the Morro Grande Forest Reserve (~9,400 ha) were classified as connected with corridors (four small and four large fragments, corridors included were 25–100 m wide native vegetation) or isolated (three small and four large fragments). Lizards were surveyed along 100 m long transects using drift fences with pitfall traps (11 traps/transect) in January–February 2002 and December 2002–January 2003 (traps open for 16 days/site, lizards individually marked prior to release).

A site comparison study in 2012 in two sites of tropical dry forest in south-western Madagascar (2) found that a site with hedges connecting different habitats had smaller differences in reptile communities than those without hedges, and that cultivated areas with hedges had more species than cultivated areas without hedges. The similarity of reptile communities in undegraded forest, degraded forest and cultivated areas was higher in the site with hedges than in the site without hedges (result reported as dissimilarity index). Nine species were found in cultivated areas with hedges (1–19 individuals) that were not found in cultivated areas with no hedges, whereas the opposite was true for only two species (1–3 individuals). Two sites were selected that contained undegraded forest, degraded forest and cultivated areas. In one site, hedges (2 m high, containing non-native *Opuntia* spp. and native vegetation) surrounded cultivated areas and bordered degraded forest. The other site had no hedges. Eight 100 m transects were established in each habitat, and all reptile species were recorded within 1.5 m of the transect line (10 surveys in February–April 2012). In cultivated areas transects followed field boundaries (hedges vs no hedges).

(1) Dixo M. & Metzger J.P. (2009) Are corridors, fragment size and forest structure important for the conservation of leaf-litter lizards in a fragmented landscape? *Oryx*, 43, 435–442.

(2) Nopper J., Lauströer B., Rödel M.O. & Ganzhorn J.U. (2017) A structurally enriched agricultural landscape maintains high reptile diversity in sub-arid south-western Madagascar. *Journal of Applied Ecology*, 54, 480–488.

12.3. Retain buffer zones around core habitat

- We found no studies that evaluated the effects of retaining buffer zones around core habitat on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background
Retaining areas of natural or semi-natural vegetation around core habitats can help to protect the habitat and wildlife that it supports from the detrimental effects of habitat loss or disturbance.

12.4. Protect specific habitat structures

- We found no studies that evaluated the effects of protecting specific habitat structures on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background
Protection of specific habitat structures (e.g. hibernacula sites) or a specific resource (e.g. food resource) may be a more effective conservation intervention than more general habitat protection especially in urban or peri-urban settings.

13. Habitat restoration and creation

Habitat destruction is the greatest threat to biodiversity worldwide and habitat protection remains one of the most important and frequently used conservation interventions. However, in many parts of the world, restoring damaged habitats, improving habitats through altering management regimes, or creating new habitat may also be possible.

Actions in this chapter include: ground vegetation and soil management (e.g. through conservation grazing), creation of habitat features (e.g. artificial refuges, ponds, and rock outcrops) and the restoration of whole ecosystems (e.g. wetlands or grasslands). For studies that discuss the effect of interventions that involve restoration through processes such as fire or fire surrogates (e.g. managing mid-storey or ground vegetation mechanically or with herbicides), see the chapter *Natural system modifications*. For studies discussing thinning or managing forests, see *Biological resource use – logging and wood harvesting*. For those that involve the control of invasive species see *Threat: invasive and other problematic species and management.*

Habitat restoration or creation is often required by law as a response to mining or other activities that destroy natural habitats. For studies that discuss the effects of restoring habitat after mining, see *Threat: energy production and mining - restore ex-mining/energy production habitat*. For studies that discuss creating habitats to offset habitat lost during development, see *Threat: residential and commercial development – create suitable habitats to offset habitats lost within development areas*.

Vegetation management

13.1. Plant native species

- **Two studies** evaluated the effects of planting native species on reptile populations. Both studies were in the USA[1,2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Occupancy/range (1 study):** One before-and-after study in the USA[2] found that an area reseeded and replanted with native vegetation was colonized and abandoned at different times by two snake and one lizard species, and one other lizard species may have remained, but in low numbers.

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One replicated, site comparison study in the USA[1] found that grasslands reseeded with both native and non-native grasses were used by Texas horned lizards.

Background

Planting vegetation can help to relatively rapidly re-establish habitats after human disturbance. This may benefit reptile populations.

Studies that combine native species planting with other restoration activities are discussed under *Whole habitat restoration*.

A replicated, site comparison study in 2001 in grassland in Texas, USA (1) found that Texas horned lizards *Phrynosoma cornutum* were present in reseeded native and non-native grassland. Texas horned lizards were observed in one reseeded native grassland plot planted without buffalo grass *Buchloe dactyloides* (one lizard), in one reseeded native grassland plot planted with buffalo grass (one lizard) and two reseeded non-native grassland plots (one lizard/plot). In July 2001, Texas horned lizards were opportunistically surveyed in 1 km^2 plots reseeded with either a native species mix excluding buffalo grass (4 plots), a native species mix including buffalo grass (4 plots), or non-native grasses (weeping lovegrass *Eragrostis curvula* or Old World bluestem *Bothriochloa ischaemum*, 7 non-native grass plots), as well as a single plot of unseeded unploughed native grass (16 total plots). Reseeded plots were part of the US Conservation Reserve Program to restore prairie.

A before-and-after study in 1999–2003 of former agricultural land in California, USA (2) found that upland habitat restored by seeding and transplanting native plant species was colonized by California king snakes *Lampropeltis getulus californiae,* western fence lizards *Sceloporus occidentalis* and gopher snakes *Pituophis catenifer*. California king snakes and western fence lizards were observed from two years after restoration took place (in 2001–2003). Gopher snakes were recorded in the year after restoration took place only. Western whiptail lizards *Cnemidophorus tigris* were recorded before restoration, but not afterwards, although the authors report that they are likely to have persisted in low numbers. In 1999, native plants were introduced to 20 plots (4 ha) in randomized blocks by either seeding or transplanting, with or without surface contouring. Visual encounter surveys (circular plots and transects) and artificial coverboard surveys (4/plot) were undertaken once before restoration in 1999 and at least 12 times thereafter in 2000–2003.

(1) McIntyre N.E. (2003) Effects of conservation reserve program seeding regime on harvester ants (*Pogonomyrmex*), with implications for the threatened Texas horned lizard (*Phrynosoma cornutum*). *Southwestern Naturalist*, 48, 274–277.

(2) Uptain C.E., Garcia K.R., Ritter N.P., Basso G., Newman D.P. & Hurlbert S.H. (2005*)* Results of a habitat restoration study on retired agricultural lands in the San Joaquin Valley, California. Pages 107–175 in: *Land Retirement Demonstration Project five year report*. US Department of the Interior, Interagency Land Retirement Team, Fresno, California.

13.2. Release animals that modify landscapes (e.g. ecological engineers)

- We found no studies that evaluated the effects on reptile populations of releasing animals that modify landscapes (e.g. ecological engineers).

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Some species modify landscapes ('ecosystem engineers') through browsing, grazing, trampling (e.g. African elephants *Loxodonta africana*) or intentionally redesigning the physical environment (e.g. European beavers *Castor fiber*). African elephants maintain open wooded grasslands through damage to trees, which increases crevice availability and possibly insect prey for the arboreal gecko *Lygodactylus keniensis* (Pringle 2008), while black-tailed prairie dog *Cynomys ludovicianus* and European rabbit *Oryctolagus cuniculus* burrows provide habitat for reptiles (Kretzer & Cully 2001, Bravo *et al*. 2009). Reintroducing ecosystem engineers where they have become locally extinct may support reptile species that depend on the habitats that these species provide.

In some situations (e.g. where species lost from landscapes have gone extinct) releasing replacement species may be the only option. For example, giant tortoises may be introduced to islands to restore habitat and ecosystem function in the place of the extinct native species (Griffiths *et al.* 2009). However, releasing non-native species is an inherently risky process, and the potential for negative consequences caused by such releases should be carefully considered beforehand.

Bravo L.G., Belliure J. & Rebollo S. (2009) European rabbits as ecosystem engineers: warrens increase lizard density and diversity. *Biodiversity and Conservation*, 18, 869–885.

Griffiths C.J., Jones C.G., Hansen D.M., Puttoo M., Tatayah R.V., Müller C.B. & Harris S. (2010) The use of extant non-indigenous tortoises as a restoration tool to replace extinct ecosystem engineers. *Restoration Ecology*, 18, 1–7.

Kretzer J.E. & Cully Jr J.F. (2001) Effects of black-tailed prairie dogs on reptiles and amphibians in Kansas shortgrass prairie. *The Southwestern Naturalist*, 171–177.

Pringle R.M. (2008) Elephants as agents of habitat creation for small vertebrates at the patch scale. *Ecology*, 89, 26–33.

13.3. Manage vegetation using livestock grazing

- **Two studies** evaluated the effects of managing vegetation using livestock grazing on reptile populations. One study was in France[1] and one was in the USA[2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Abundance (1 study):** One site comparison study in France[1] found that one reptile species was more abundant in areas grazed by sheep than in areas managed by burning, whereas the abundance of five other species was similar in all areas.
- **Reproductive success (1 study):** One before-and-after study in the USA[2] found that after grazing cattle to restore bog turtle habitat, along with providing artificial nest covers, more bog turtle eggs were laid and hatching success was higher than before.

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One before-and-after study in the USA[2] found that bog turtle nests were laid only in areas that had been grazed in the current or previous growing season.

Background

Using grazing to manage vegetation can limit succession that would otherwise lead to an increase in woody plant species and may create microhabitat diversity through trampling. This may help to increase the abundance of reptile species that depend on early-succession and diverse habitats.

Here we present studies that use grazing to target both native and non-native species.

For studies discussing the impacts of different grazing regimes in agricultural environments, see *Threat: Agriculture – manage grazing regime* and *Threat: agriculture – cease livestock grazing*.

For studies discussing the effects of managing vegetation by cutting, using herbicide or by hand, see *Manage vegetation by cutting or mowing*; *Manage vegetation using herbicides* and *Manage vegetation by hand (selective weeding).*

A site comparison study in 2016 in an area of heathland in Nouvelle-Aquitaine, France (1) found that one of six reptile species was more abundant in a site grazed by sheep than in sites that were burned 5–12 years previously, whereas the other five species were similarly abundant across all sites. More western green lizards *Lacerta bilineata* were found in the grazed area (1.5 lizards/site) than in any of the burned areas (0.1 lizards/site for all burned sites), whereas no difference was found between grazed or burned areas in the number of wall lizards *Podarcis muralis* (0–4 lizards/site) or the number of four snakes species (green whip snake *Hierophis viridiflavus*, viperine snake *Natrix maura*, grass snake *Natrix natrix* and European asp *Vipera aspis*; data not presented). An area of heathland (135 ha) was managed by annual sheep grazing or prescribed burning. One grazed site and three burned sites (all sites 8–10 ha) were selected (one each burned 5, 10 or 12 years ago). In 2016, a total of 96 cover boards (corrugated roofing tiles) were split between the four areas (24 boards/area), and 10 surveys were conducted in April–June. Reptiles found on or under cover boards were counted.

A before-and-after study in 2009–2016 in wet meadow and marsh in New York State, USA (2) found that after grazing cattle to restore bog turtle *Glyptemys muhlenbergii* habitat along with using artificial nest covers, more eggs were laid in more nests, hatching rates increased and more juveniles were observed. Results were not statistically tested. More bog turtle eggs were laid after grazing commenced and artificial nest covers were used (2012–2016: 15–47 eggs in 3–12 nests/year) compared to before (2009–2010: 7–8 eggs in 2–3 nests/year). After grazing started, all nests were found in areas that had been grazed in the current or previous growing season. Overall hatching success was 52% (58 of 112 eggs hatched) compared to 27% when nests were not protected and there was no grazing (4 of 15 eggs hatched). More juveniles were observed at the end of the grazing program (2016: 6 juveniles/year) compared to at the start (2012: 1 juvenile/year). In 2012–2014 and 2016, one or both of two adjacent fenced paddocks (3.6 ha total area, 1.6 ha of bog turtle habitat) were grazed by 0.6–1.4 cattle/ha for 5–21 weeks in April-October (see original paper for details) and bog turtle nests were protected by mesh-cloth artificial nest covers (12 x 12 x 12 cm). In 2009–2010, there was no grazing and bog turtle nests were not protected. Nests were located by surveying on foot in 2009–2016 (2014 data were excluded). Turtles were monitored by radio tracking and on foot observations in 2012–2016.

(1) Pernat A., Sellier Y., Préau C. & Beaune D. (2017) Effet du pâturage sur le lézard vert occidental (*Lacerta bilineata* Daudin, 1802) (Squamata: Lacertidae) en milieu de landes. *Bulletin de la Société Herpétologique de France*, 161, 57–66.
(2) Travis K.B., Kiviat E., Tesauro J., Stickle L., Fadden M., Steckler V. & Lukas L. (2018) Grazing for bog turtle (*Glyptemys muhlenbergii*) habitat management: Case study of a New York fen. *Herpetological Conservation and Biology*, 13, 726–742.

13.4. Manage vegetation using herbicides

- **Seven studies** evaluated the effects of managing vegetation using herbicides on reptile populations. Four studies were in the USA[1,2,5,6], two were in Australia[4,7] and one was in the US Virgin Islands[3].

COMMUNITY RESPONSE (3 STUDIES)

- **Community composition (1 study):** One replicated, randomized, controlled, before-and-after study in Australia[4] found that areas where an invasive shrub was sprayed with herbicide had similar composition of reptile species compared to unsprayed areas.
- **Richness/diversity (3 studies):** Three replicated, randomized, controlled, before-and-after studies in Australia[4] and the USA[5,6] found that areas where vegetation was treated with herbicides had similar richness of reptile species[4,6] or combined reptile and amphibian species[5] compared to areas not treated with herbicide.

POPULATION RESPONSE (6 STUDIES)

- **Abundance (4 studies):** Three of four studies (including three replicated, randomized, controlled, before-and-after studies) in the USA[2,5,6] and Australia[4] found that areas where vegetation was treated with herbicides had similar abundance of all[4,6] or most[5] reptiles compared to areas not treated with herbicide. The other study[2] found that after glyphosate was applied to pond vegetation, fewer mangrove salt marsh snakes were found compared to immediately before application.
- **Reproductive success (2 studies):** One of two controlled studies (including one replicated study and one randomized study) in the USA[1] and US Virgin Islands[3] found that exposure of red-eared slider eggs to high levels of glyphosate caused a reduction in hatching success[1]. The other study[3] found that leatherback turtle nests in areas treated with herbicide had similar hatching and emergence success compared to nests in untreated areas.
- **Survival (1 studies):** One randomized, controlled study in the US Virgin Islands[3] found that in areas treated with herbicide, fewer leatherback turtle hatchlings became entangled in vegetation than in untreated areas.

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One replicated, randomized, controlled study in Australia[7] found that pink-tailed worm-lizards were not found in restored rocky areas treated with herbicide, but were found in restored areas not treated with herbicide.

Background

Herbicides can be used as a substitute for prescribed fire to eliminate competing mid-story or ground vegetation. Although herbicides do not have the multiple ecosystem functions provided by fire, they have some advantages such as increased selectivity and decreased risk of offsite fire damage.

Here we present studies that remove both native and non-native species. Studies that investigate the effects of removing vegetation as one of a combination of restoration management actions are discussed in the actions under the subsection *Whole habitat restoration*.

For studies that use herbicide in combination with prescribed burning, see *Threat: Natural system modifications – Use prescribed burning in combination with herbicide application*.

For studies discussing the effects of grazing as part of habitat restoration, see *Manage vegetation using livestock grazing*. For studies discussing the effects of managing vegetation by cutting or by hand, see *Manage vegetation by cutting or mowing* and *Manage vegetation by hand (selective weeding).*

A replicated, controlled study in 2005 in a laboratory in Louisiana, USA (1) found that the herbicide (glyphosate), which may be used to manage vegetation, when applied to eggs of red-eared sliders *Trachemys scripta elegans,* reduced hatching success and the health of hatchlings, but only at the highest glyphosate concentration. Hatching success at the highest concentration of glyphosate and the surface-active agent was 73%, compared to hatching success of 80–100% with lower concentrations or no glyphosate. Hatchlings from eggs that had been exposed to the highest concentration of glyphosate and surface-active agent also weighed less both at hatching and after 14 days, compared to those from eggs that had been exposed to lower concentrations. Eggs of red-eared terrapins were exposed to either a single application of glyphosate (68–11,206 ppm wet weight of glyphosate in Glypro, 15 eggs/concentration) with a surface-active agent (LI700) or to no glyphosate (16 eggs). After exposure, eggs were incubated in vermiculite and the number to hatch successfully was recorded. Fourteen days after hatching, size and weight of hatchlings was recorded.

A before-and-after study in 2006–2007 in two connected stormwater run-off ponds in Florida, USA (2) found that after glyphosate herbicide was applied in summer to remove ground vegetation, fewer mangrove salt marsh snakes *Nerodia clarkia compressicauda* tended to be recorded in the autumn. Results were not statistically tested. In the autumn after glyphosate herbicide was applied to pond vegetation, mangrove salt marsh snake abundance was estimated as 47 snakes, compared to 95 snakes in spring immediately before herbicide was applied, 94 snakes in the summer while herbicide was being applied and 33 snakes in the spring of the year prior to herbicide being applied. Two man-made ponds (0.2–0.4 ha) were treated with glyphosate herbicide (‘Aquamaster’) monthly during summer 2007 (exact start date not known). Salt marsh snakes were monitored at night for three nights at a time in April 2006 (spring), March–April 2007 (spring), May–July 2007 (summer) and August–October 2007 (autumn). Snakes were caught by hand, individually marked with PIT tags and released. Snakes >40 cm long were used to calculate abundance.

A randomized, controlled study in 2007 on coastal sand dunes in St Croix, US Virgin Islands (3) found that using herbicide to remove the native plant ‘beach morning glory’ *Ipomoea pes-caprae* did not increase leatherback turtle

Dermochelys coriacea nest productivity compared to plots with untreated vegetation, but fewer hatchlings became entangled in plant roots in herbicide treated plots compared to in untreated, vegetated plots. Herbicide treated plots had similar hatching success (24% hatched/total yolked eggs) and emergence success (21% hatched/total yolked eggs) compared to untreated vegetated plots (hatching: 20%; emergence: 15%), but lower hatching and emergence success compared to naturally unvegetated plots (hatching: 50%; emergence: 38%). However, the number of hatchlings that became entangled in plant roots was lower in herbicide-treated plots (17 of 393, 4% of hatchlings trapped) compared to untreated vegetated plots (36 of 314, 12% of hatchlings trapped). Ten herbicide-treated plots and 10 untreated vegetation plots (5 x 5 m) were randomly allocated across two experimental blocks. Ten unvegetated plots were established seaward of the experimental blocks. Herbicide (3% Roundup Pro ConcentrateTM) was applied once, 72 days before the nesting season. In April–May 2007, nests laid in areas of the beach liable to flooding were relocated to either herbicide-treated plots, untreated vegetated plots or naturally unvegetated plots (2 nests/plot; 16–20 nests/treatment). Nylon mesh nets were placed over nests before hatching. All nests were excavated 1–3 days after the main period of hatchling emergence and the number of hatched and unhatched eggs was counted.

A replicated, randomized, controlled, before-and-after study in 2010–2012 in shrubland in New South Wales, Australia (4) found that spraying invasive Bitou bush *Chrysanthemoides monilifera* ssp. *rotundata* with herbicide did not increase reptile abundance or species richness in the year after spraying. Reptile abundance and species richness was similar after shrubland was sprayed (0.4–1.0 individuals/100 m^2; 0.4–0.5 species/100 $m^{2,}$ respectively) compared to before spraying (0.6 individuals/100 m^2; 0.5 species/100 m^2) and compared to sites where Bitou bush was unsprayed (0.9–1.0 individuals/100 m^2; 0.3–0.5 species/100 m^2) and unsprayed sites without Bitou bush (0.6–1.3 individuals/100 m^2; 0.3 species/100 m^2). Species composition was similar before-and-after spraying and between sprayed and unsprayed sites. Reptiles were surveyed in 10 sites in March–April 2010, November 2010, and February 2011. Two sites contained invasive Bitou bush and were treated with glyphosate herbicide in May–June 2010. Eight sites were not sprayed: three contained invasive Bitou bush and five did not. Where Bitou bush was present, it comprised 40% cover in a mosaic with native vegetation. Reptiles were surveyed morning and evening (15 minutes/transect) using active searches (for example, turning over logs and rocks, raking leaf litter, lifting loose bark).

A replicated, randomized, controlled, before-and-after study in 1999–2007 in six pine plantations in Mississippi, USA (5) found that herbicide application did not increase reptile and amphibian diversity in six out of seven years of monitoring, although eastern fence lizards *Sceloporus undulatus* abundance did increase in the year after management. In six of seven years after herbicide application, species richness, diversity measures and most species abundances were similar in plots treated with herbicide and plots receiving no treatment (data reported as statistical model outputs, see paper for details). Eastern fence lizard abundance was higher in plots treated with either herbicide only, herbicide in combination with burning or burning alone (0.02 lizards/plot; abundance for herbicide only plots not provided separately) in the first year after management compared to unmanaged plots (0.002 lizards/plot; abundance in other years not provided). Four 10 ha plots were set up in six intensively managed 18–22-year-old commercial pine stands (59–120 ha). Plots were either treated with herbicide ('Imazapyr') in September 1999, burned in the dormant season (December–February) in 2000, 2003 and 2006; or treated with herbicide then burned; or unmanaged. Reptiles were monitored using drift fences with pitfall and funnel traps in May–June 1999–2007 (one year before management and seven years after management began).

A replicated, randomized, controlled, before-and-after study in 2008–2014 in an upland mixed oak forest in the Appalachians, USA (6) found that mid-storey vegetation removal using herbicides did not increase the abundance or species richness of total reptiles, snakes or lizards when compared to no management. Abundance and richness of total reptiles was similar in herbicide treated plots (Abundance: 0.4–0.7 average captures/100 fence nights; richness of total reptiles provided as statistical model result) compared to unmanaged plots (Abundance: 0.2–0.5 average captures/100 fence nights). In 2008, mid-storey vegetation removal was carried out using herbicides. Reptiles were surveyed in herbicide-managed plots and unmanaged plots (4–5 plots/type each 225 x 225 m) using drift fences, pitfall and funnel traps in May–August one year pre-management (in 2008) and up to 5 years after management (in 2010, 2011, 2013, 2014).

A replicated, randomized, controlled study in 2014–2015 in six rock and grassland areas in Australian Capital Territory, Australia (7) found that pink-tailed worm-lizards *Aprasia parapulchella* did not recolonise restored rocky areas replanted with grasses that were treated with herbicide, but did recolonise restored areas not treated with herbicide. When herbicide was used following native grass and rock cover restoration, pink-tailed worm-lizards did not recolonise rocks (no lizards observed), but four lizards and one lizard skin were observed in restored rock outcrops not treated with herbicide. There was no evidence of lizards in unrestored sites in poor quality habitat, but four lizards and three shed skins were observed in unrestored sites near high quality lizard habitat. In April–May 2014, plots (4 x 4 m) in six sites (150 m apart) were managed with: rock addition (30% rock cover) and native grass restoration; or rock addition and grass restoration with herbicide application (Glyphosate, 1:100 glyphosate:water). In February 2015, all plots were surveyed for lizards (live sightings and skins) including two unmanaged plots/site (one in poor, the other near high-quality lizard habitat).

(1) Sparling D., Matson C., Bickham J. & Doelling-Brown P. (2006) Toxicity of glyphosate as glypro and LI700 to red-eared slider (*Trachemys scripta elegans*) embryos and early hatchlings. *Environmental Toxicology and Chemistry*, 25, 2768–2774.

(2) Ackley J.W. & Meylan P.A. (2010) Watersnake eden: Use of stormwater retention ponds by mangrove salt marsh snakes (*Nerodia clarkii compressicauda*) in urban Florida. *Herpetological Conservation and Biology*, 5, 17–22.

(3) Conrad J.R., Wyneken J., Garner J.A. & Garner S. (2011) Experimental study of dune vegetation impact and control on leatherback sea turtle *Dermochelys coriacea* nests. *Endangered Species Research*, 15, 13–27.

(4) Martin L.J. & Murray B.R. (2013) A preliminary assessment of the response of a native reptile assemblage to spot-spraying invasive Bitou Bush with glyphosate herbicide. *Ecological Management and Restoration*, 14, 59–62.

(5) Iglay R.B., Leopold B.D. & Miller D.A. (2014) Summer herpetofaunal response to prescribed fire and herbicide in intensively managed, mid-rotation pine stands in Mississippi. *Wildlife Society Bulletin*, 38, 33–42.
(6) Greenberg C.H., Moorman C.E., Raybuck A.L., Sundol C., Keyser T.L., Bush J., Simon D.M. & Warburton G.S. (2016) Reptile and amphibian response to oak regeneration treatments in productive southern Appalachian hardwood forest. *Forest Ecology and Management*, 377, 139–149.
(7) McDougall A., Milner R.N.C., Driscoll D.A. & Smith A.L. (2016) Restoration rocks: integrating abiotic and biotic habitat restoration to conserve threatened species and reduce fire fuel load. *Biodiversity and Conservation*, 25, 1529–1542.

13.5. Manage vegetation by cutting or mowing

- **Seven studies** evaluated the effects of managing vegetation by cutting or mowing on reptile populations. Five studies were in the USA[2-4,6,7], one was in Australia[1], and one was in Spain[5].

COMMUNITY RESPONSE (3 STUDIES)

- **Richness/diversity (3 studies):** Three replicated, randomized, controlled studies (including one before-and-after study) in the USA[2,4,7] found that areas where vegetation was managed by cutting had similar reptile species richness compared to areas with no cutting.

POPULATION RESPONSE (5 STUDIES)

- **Abundance (4 studies):** Three of four replicated, controlled studies (including three randomized studies) in the USA[2,4,6,7] found that areas where vegetation was managed by cutting had similar reptile abundance compared to areas with no cutting[2,4,7]. The other study[6] found that densities of eastern Massassauga rattlesnakes were higher after cutting during the first three years, but similar after four years.
- **Condition (1 study):** One replicated, randomized, controlled, before-and-after study in Spain[5] found that an adapted brush cutter caused less damage to tortoise carcasses than a conventional brush cutter.

BEHAVIOUR (1 STUDIES)

- **Use (2 studies):** One replicated, randomized, controlled study in Australia[1] found that after cutting back canopy vegetation, reptiles used unshaded rocks more often than shaded rocks in winter but not spring. One randomized study in the USA[3] found that mown areas were used for nesting by Blanding's turtles less frequently than tilled areas.

Background

Vegetation may be managed by cutting or mowing as part of habitat restoration to reduce shading or prevent natural succession where specific habitat types are desired, or where invasive species are out-competing native species. A range of tools may be used for such management, from machetes and billhooks, to chainsaws, strimmers and brush cutters.

Removing vegetation may also be used as a substitute for prescribed fire to eliminate competing mid-storey or ground vegetation. Although this technique does not have the multiple ecosystem functions provided by fire, it has advantages, such as increased selectivity and decreased risk of offsite fire damage.

Here we present studies that remove both native and non-native species, ranging from grasses and other herbaceous plants, to woody shrubs and canopy vegetation. Studies that investigate the effects of removing vegetation as one of a combination of restoration management actions are discussed in the actions under the subsection *Whole habitat restoration*.

For studies that combine cutting or mowing vegetation with prescribed burning, see *Threat: Natural system modifications – Use prescribed burning in combination with vegetation cutting*.

For studies discussing the effects of grazing as part of habitat restoration, see *Manage vegetation using livestock grazing*. For studies discussing the effects of managing vegetation by hand or using herbicides, see *Manage vegetation by hand (selective weeding)* and *Manage vegetation using herbicide.*

A replicated, randomized, controlled study in 2002–2003 of rock outcrops on a sandstone plateau in New South Wales, Australia (1) found that after removing overhanging canopy, reptiles used unshaded rocks more often in winter, but not in spring, than shaded rocks. After canopy removal, reptile use of unshaded rocks was similar to shaded rocks in spring (unshaded: 38% of rocks used, shaded: 12%) but higher in winter (unshaded: 88%, shaded: 0%). Two broad-headed snakes *Hoplocephalus bungaroides*, four Lesueur's velvet gecko *Oedura lesueurii*, and one red-throated skink *Acritoscincus platynotum* were recorded under unshaded rocks and one Lesueur's velvet gecko under a shaded rock. In May 2002, sixteen rocks in three sites shaded by emerging shrubs and saplings were managed either by increasing canopy openness by 15% ('unshaded') or unmanaged ('shaded', 8 rocks/management type). Reptiles were sampled in June and August 2003 by searching under rocks and individually marking captured reptiles.

A replicated, randomized, controlled, before-and-after study in 2001–2004 in an upland hardwood forest in North Carolina, USA (2; same experimental set-up as 7) found that mechanically cutting understory vegetation did not increase overall reptile abundance or species richness. Total reptile relative abundance and species richness were both similar after mechanical understory removal (abundance in 2002: 8–13 reptiles/100 nights; richness in 2002: 4–6 species) compared to the year before it took place (abundance in 2001: 4–8 reptiles/100 nights; richness in 2001: 2 species). Reptile abundance increased 40% in all sites (including those with no management) between the two years, but this was not related to understory removal (see original paper for details). Three forest segments were divided into management zones (14 ha each): mechanical vegetation removal and no management. Chainsaws were used to remove understory vegetation in winter 2001–2002. Reptiles were surveyed using drift fences with pitfall and funnel traps before any management took place in August–October 2001 and after management in May–September 2002–2004. In total 13 reptile species were caught.

A randomized study in 2006–2008 in wetlands in New York, USA (3) found fewer female Blanding's turtle *Emydoidea blandingii* nested in mown plots than in tilled plots. Overall, fewer turtles nested in mowed plots (two turtles in 2006) that in tilled plots (7 turtles in 2006; 5 turtles in 2008). In 2006, nine of 10 monitored female turtles nested, and in 2008, six turtles nested. Two turtles nested in the same physical plot each year, in spite of a change in management. In 2006, thread trailing revealed that all female turtles explored or had been placed on each plot type before choosing where to nest. Eight sites around the edge of a fenced 12 ha wetland were monitored for turtle nesting activity. Two plots (5 x 7 m each) were established at each site, and one/site was either mowed to 5 cm height or tilled to a depth of 15 cm (treatment randomly applied in 2006 and 2008). Nesting activity was monitored by visual searches and radio tracking or by attaching a bobbin and thread to female turtles in May and June 2006 and 2008 (10 turtles monitored in total).

A replicated, randomized, controlled study in 2006–2007 in hardwood forests in North Carolina, USA (4) found overall reptile species richness and capture rates were similar in areas with vegetation cutting compared to areas with no cutting. Overall reptile richness and overall reptile, snake, lizard and turtle captures were similar after vegetation cutting (richness: 6–7 species/100 trapping nights, overall captures: 6 individuals/100 trapping nights, snakes: 1–2 individuals/100 trapping nights, lizards: 4–5 individuals/100 trapping nights, turtles: 0 individuals/100 trapping nights) and no cutting (richness: 6 species, overall captures: 7–7 individuals, snakes: 3–5 individuals, lizards: 4 individuals, turtles: 0 individuals). Three blocks of four sets of 10 ha sites were either managed by cutting vegetation (using chainsaws to cut trees and understory, 2001–2002) or were left uncut. Reptiles were surveyed in May–August 2006 and 2007 using a group of drift fences with pitfall traps (3 groups/site).

A replicated, randomized, controlled study in 2013 in abandoned vineyards, pine and oak forest in Catalonia, Spain (5) found that a specially adapted brush cutter accessory minimised tortoise cutting injuries during vegetation cutting. When an adapted brush cutter was used, no tortoise carcasses were damaged, but when a conventional brush cutter was used, the majority of tortoise carcasses sustained what would have been fatal injuries (yearlings: 40% no damage, 60% fatal wounds; juveniles: 60% serious damage, 40% fatal wounds; subadults and adults: 100% fatal wounds). In February 2013, eight plots (100m^2 each) were cleared of shrub cover using either a modified brush cutter (6 plots) or conventional cutter (2 plots). One-hundred and four frozen tortoises (5 yearlings, 5 juveniles and 3 adults/plot) were randomly distributed under shrubs and the impact on tortoise carcasses was assessed immediately after cutting.

A replicated, controlled study in 2008–2012 in swamp forest and shrubland in New York State, USA (6) found that where shrubs and canopy cover were reduced, densities of eastern Massassauga rattlesnakes *Sistrurus catenatus* were higher in the first three years after cutting, but densities in cut and uncut plots were similar after four years. The effect of removing canopy or shrubs cannot be separated. Estimated rattlesnake densities were greater in 0-year-old (0.072–0.141 snakes/100 m^2), 1-year-old (0.045 snakes/100 m^2) and 3-year-old (0.133 snakes/100 m^2) cut plots than uncut plots (0.003–0.009 snakes/100 m^2). Rattlesnake densities in 4-year-old cut plots (0.013 snakes/100 m^2) were similar to uncut plots. Canopy was reduced by cutting shrubs to <0.25 m high in 50 plots in two known rattlesnake breeding areas in 2008 (six 28 m^2 plots), 2011 (thirty-two 100 m^2 plots) and 2012 (twelve 28 m^2 plots). In addition, 4 ha of adjacent forest was mechanically cleared in 2011. Snakes were monitored using visual encounter surveys in 66 locations with canopy removal (50 plots of cut vegetation in breeding areas and sixteen 36 m^2 plots within the forest areas cleared in 2011), and 44 areas with no vegetation removal (twenty-eight 28 m^2 plots of uncut vegetation in breeding areas and sixteen 36 m^2 plots in uncut forest). In 2011 and 2012, the number of snakes caught in canopy removal areas (removal having occurred 0–4 years previously) was compared to the number of snakes in uncut plots. It is unclear whether the results reported are based on the breeding areas only or include the cut and uncut forest plots. Surveys were carried out once a week in June–August 2011 and May–August 2012. Snakes were captured, sexed and individually marked with PIT tags.

A replicated, randomized, controlled study in 2001–2016 in upland forest in North Carolina, USA (7, same experimental set-up as 2) found that cutting vegetation did not increase overall reptile species richness or the abundance of different species compared to areas with no cutting. Overall reptile species richness was similar in areas with vegetation cutting and in areas with no cutting (data reported as statistical model outputs, see original paper for details). Eastern fence lizard *Sceloporus undulatus* and five-lined skink *Plestiodon fasciatus* abundance was similar in cut (1–3 individuals/100 trapping nights) and uncut (0–2 individuals/100 trapping nights) plots, and all other reptile species were excluded from analysis due to small sample sizes (20 species total, see paper for details). Three similar study sites were selected within a 5,841ha mixed oak-hickory forest. Within each site, one plot each (10 ha core areas with 20 m wide buffers) was managed by cutting understorey vegetation (in winters 2001–2002 and 2011–2012) or was left uncut. Reptiles were surveyed after management using drift fences with pitfall and funnel traps ('arrays') in May–August of 2003–2004, 2006–2007, and 2014–2016 (158–341 array nights/plot/year).

(1) Webb J.K., Shine R. & Pringle R.M. (2005) Canopy removal restores habitat quality for an endangered snake in a fire suppressed landscape. *Copeia,* 2005, 894–900.

(2) Greenberg C.H. & Waldrop T.A. (2008) Short-term response of reptiles and amphibians to prescribed fire and mechanical fuel reduction in a southern Appalachian upland hardwood forest. *Forest Ecology and Management*, 255, 2883–2893.

(3) Dowling Z., Hartwig T., Kiviat E. & Keesing F. (2010) Experimental management of nesting habitat for the Blanding's turtle (*Emydoidea blandingii*). *Ecological Restoration*, 28, 154–159.

(4) Matthews C.E., Moorman C.E., Greenberg C.H. & Waldrop T.A. (2010) Response of reptiles and amphibians to repeated fuel reduction treatments. *The Journal of Wildlife Management*, 74, 1301–1310.

(5) Vilardell-Bartino A., Capalleras X., Budo J., Bosch R. & Pons P. (2015) Knowledge of habitat preferences applied to habitat management: the case of an endangered tortoise population. *Amphibia-Reptilia*, 36, 13–25.

(6) Johnson B.D., Gibbs J.P., Bell T.A. & Shoemaker K.T. (2016) Manipulation of basking sites for endangered eastern massasauga rattlesnakes. *The Journal of Wildlife Management*, 80, 803–811.

(7) Greenberg C.H., Moorman C.E., Matthews-Snoberger C.E., Waldrop T.A., Simon D., Heh A. & Hagan D. (2018) Long-term herpetofaunal response to repeated fuel reduction treatments. *Journal of Wildlife Management*, 82, 553–565.

13.6. Manage vegetation by hand (selective weeding)

- **Four studies** evaluated the effects of managing vegetation by hand on reptile populations. Two studies were in the USA[2,3], one was in South Africa[1], and one was in the US Virgin Islands[4].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Abundance (1 study):** One replicated, paired sites, controlled, before-and-after study in the USA[2] found that removing invasive, non-native Sahara mustard by hand had mixed effects on the abundance of two lizard species.
- **Reproductive success (1 study):** One replicated, randomized, controlled study in the US Virgin Islands[4] found that in areas where native beach morning glory was removed by hand, leatherback turtle nests had similar hatching and emergence success compared to areas where no removal took place.
- **Survival (1 study):** One replicated, randomized, controlled study in the US Virgin Islands[4] found that in areas where native beach morning glory was removed by hand, fewer leatherback turtle hatchlings became entangled in vegetation compared to areas where no removal took place.

BEHAVIOUR (2 STUDIES)

- **Use (2 studies):** One replicated, controlled, before-and-after study in South Africa[1] found that removing an invasive plant by hand resulted in more sites being used for nesting by Nile crocodiles compared to areas with no removal. One randomized study in the USA[3] found that weeded or mown areas were used less frequently for nesting by Blanding's turtles than tilled areas.

Background

Many habitats depend on disturbances such as grazing or fire to reduce succession that leads to increases in woody plant species and conversion to forest. Manually removing vegetation may prevent succession and reduce shading.

Here we present studies that remove both native and non-native species.

For studies discussing the effects of grazing as part of habitat restoration, see *Manage vegetation using livestock grazing*. For studies discussing the effects of managing vegetation by cutting or using herbicides, see *Manage vegetation by cutting or mowing* and *Manage vegetation using herbicide.*

A replicated, controlled, before-and-after study in 1993–1997 in shoreline habitat on a lake in Kwazulu-Natal, South Africa (1) found that removing the invasive plant *Chromolaena odorata* from nesting sites by hand increased Nile crocodile *Crocodylus niloticus* successful nesting attempts over three breeding seasons. Results were not statistically tested. Known nesting sites where invasive vegetation was removed had 40% (2 out of 5 sites nested), 80% and 60% success over three breeding seasons following removal, compared to 40% nesting success before removal. Newly created nesting sites, where invasive vegetation was completely removed, had 33% (2 out of 6 sites nested), 33% and 67% success over three breeding seasons following removal, compared to 0% success before removal. Nesting success in sites where invasive vegetation was not removed was 100% (5 out of 5 sites nested), 60%, 40% and 40% over four breeding seasons. In 1993, sixteen nest sites were chosen: five known nesting sites where the invasive plant was present and manually removed from 1994; six sites newly created by manually removing all invasive vegetation and root stock (4 x 4 m area); and five where the invasive plant was present and was not removed. In 1994–1997 (three breeding seasons) invasive vegetation clearing was carried out each season. In 1993–1997, all sites were monitored using foot, boat and aerial surveys in mid-December to determine use of nesting sites.

A replicated, paired sites, controlled, before-and-after study in 2002–2007 in a site of dunes and desert scrub in California, USA (2) found that manual removal of invasive non-native Sahara mustard *Brassica tournefortii* resulted in an increase in Coachella Valley fringe-toed lizard *Uma inornata* abundance compared to no weeding in one of three years in one of two habitat types, but flat-tailed horned lizard *Phrynosoma mcallii* abundance remained similar in all comparisons. In yearly comparisons, fringe-toed lizard abundance was higher in weeded plots in one of three years during or after weeding in active dunes (second year of weeding: 7 lizards/plot; not weeded: 4 lizards/plot; first & third years: 2–4 lizards/plot), but not in stabilized sand fields (weeded: 2 lizards/dune; not weeded: 1–3 lizards/dune). Overall abundance of fringe-toed lizards was higher in weeded plots (3 lizards/plot) compared to plots with no weeding (2 lizards/plot), but flat-tailed horned lizard abundance was similar in both (weeded: 0.1 lizards/plot; not weeded: 0.1 lizards/plot). Paired plots (10 x 100 m plots) of mustard removal and no mustard removal were established in stabilised sand fields (15 removal plots, 15 no removal plots) and active dunes (6 removal plots, 6 no removal plots). Mustard removal was carried out by hand in 2005–2006. Reptiles were surveyed at each site six times/year from May to July 2002–2007 in the morning using sightings and tracks left in the sand.

A randomized study in 2006–2008 in wetlands in New York, USA (3) found that female Blanding's turtle *Emydoidea blandingii* used weeded plots less frequently than tilled plots for nesting. Overall, fewer turtles nested in weeded plots (1 turtle in 2008) than in tilled plots (7 turtles in 2006; 5 turtles in 2008). In 2006, nine of 10 monitored female turtles nested, and in 2008, six turtles nested. Two turtles nested in the same physical plot each year, in spite of a change in management. In 2006, thread trailing revealed that all female turtles explored or had been placed on each plot type before choosing where to nest. Eight sites around the edge of a fenced 12 ha wetland were monitored for turtle nesting activity. Two plots (5 x 7 m each) were established at each site, and one/site was either hand weeded (90% of vegetation removed) or tilled to a depth of 15 cm (treatment randomly applied in 2006 and 2008). Nesting activity was monitored by visual searches and radio tracking or by attaching a bobbin and thread to female turtles in May and June 2006 and 2008 (10 turtles monitored in total).

A replicated, randomized, controlled study in 2007 on coastal sand dunes in St Croix, US Virgin Islands (4) found that manually removing the native plant 'beach morning glory' *Ipomoea pes-caprae* did not increase leatherback turtle *Dermochelys coriacea* nest productivity compared to plots with unmanaged vegetation, but fewer hatchlings became entangled in plant roots in removal plots compared to in untreated, vegetated plots. Manually-removed vegetation plots had similar hatching success (24% hatched/total yolked eggs) and emergence success (emergence success: 20% hatched/total yolked eggs) compared to unmanaged vegetated plots (hatching success: 20%; emergence success: 14% hatched/total yolked eggs), but lower hatching and emergence success compared to naturally non-vegetated plots (hatching success: 50%; emergence success: 38% hatched/total yolked eggs). However, the number of hatchlings that became entangled in plant roots was lower in plots with vegetation removed (19 of 397, 5% hatchlings trapped) compared to unmanaged vegetated plots (36 of 314, 11% hatchlings trapped). Vegetation removal was carried out using machetes and weeding by hand until no vines remained above ground in April before the start of the nesting season. Ten manually-removed and 10 unmanaged vegetation plots (5 x 5 m) were randomly allocated across two experimental blocks. Ten unvegetated plots were established seaward of the experimental blocks. In April–May 2007, nests laid in areas of the beach liable to flooding were relocated to either plots with manually-removed vegetation, untreated vegetated plots or naturally unvegetated plots (2 nests/plot; 16–20 nests/treatment). Nylon mesh nets were placed over nests before hatching. All nests were excavated 1–3 days after the main period of hatchling emergence and the number of hatched and unhatched eggs was counted.

(1) Leslie A.J. & Spotila J.R. (2001) Alien plant threatens Nile crocodile (*Crocodylus niloticus*) breeding in Lake St. Lucia, South Africa. *Biological Conservation,* 98, 347–355.
(2) Barrows C.W., Allen E.B., Brooks M.L. & Allen M.F. (2009) Effects of an invasive plant on a desert sand dune landscape. *Biological Invasions*, 11, 673–686.
(3) Dowling Z., Hartwig T., Kiviat E. & Keesing F. (2010) Experimental management of nesting habitat for the Blanding's turtle (*Emydoidea blandingii*). *Ecological Restoration*, 28, 154–159.
(4) Conrad J.R., Wyneken J., Garner J.A. & Garner S. (2011) Experimental study of dune vegetation impact and control on leatherback sea turtle *Dermochelys coriacea* nests. *Endangered Species Research*, 15, 13–27.

13.7. Clear or open patches in forests

- **Six studies** evaluated the effects of removing canopy to create clearings on reptile populations. Two studies were in the USA[5,6] and one was in each of Sweden[1], Australia[2], the UK[3] and France[4].

COMMUNITY RESPONSE (1 STUDY)

- **Richness/diversity (1 study):** One replicated, controlled study in Australia[2] found that rocky outcrops where trees were removed had higher reptile species richness than overgrown outcrops, and similar richness to outcrops that were naturally sun exposed.

POPULATION RESPONSE (5 STUDIES)

- **Abundance (4 studies):** One of four replicated studies (including three controlled studies) in Sweden[1], Australia[2], the UK[3] and the USA[5] found that after clearings and sand patches were created, sand lizard colonized, abundance then declined, but then increased once more, larger clearings were created[1]. One study[3] found that more slow worms and common lizards were found in open areas of woodland maintained by vegetation cutting compared to in coppiced areas. One study[5] found that areas with reduced canopy had more eastern Massassauga rattlesnakes in the first three years after cutting than uncut areas, but similar numbers after four years. The other study[2] found that removing trees from rocky outcrops had mixed effects on reptile abundance.

- **Occupancy (1 study):** One replicated, controlled, before-and-after study in France[4] found that forest areas where the canopy had been opened up were more likely to be occupied by asper vipers than areas with closed canopy.

BEHAVIOUR (1 STUDY)

- **Behaviour change (1 study):** One before-and-after study in the USA[6] found that clearing a patch of canopy in a forest did not affect spotted turtle home range size.

Background

Clearings in forests and woodland can be natural features that add diversity to the habitat. They can be created by natural events, such as mature trees falling, and maintained by grazing animals. In absence of natural clearings (such as in a younger forest) artificially removing canopy vegetation may mimic the same conditions.

A replicated, controlled, before-and-after study in 1988–2004 in sandy pine heath in south-central Sweden (1) found that in areas where trees were cleared and sand patches created, sand lizards *Lacerta agilis* colonized, abundance then declined, but then increased after more clearings were created. Sand lizards gradually colonized the newly created clearings and eventually abandoned unmanaged habitat after 16 years (see original paper). During the first 10 years after clearing and sand patch creation, female sand lizard abundance declined (9–11%) but increased annually (11–19%) after a second, larger-scale clearing and sand patch creation programme was carried out. In particular, subadult lizard abundance increased more after the second creation programme (after first programme <10% increase; after second programme >150% increase in relative population size). Restoration was first carried out in 1988 and 1992 when nine 1–2 ha lizard-appropriate habitat patches in two sites were managed by tree felling and creating sand patches. A second restoration programme took place in the same two sites in 1999 and 2001, creating 18 habitat patches (10 ha each) by tree felling, soil scarification and excavating 7–11 sand patches/site (100–200 m^2 patches). Sand lizards were monitored from May–September in 1988–2004 by hand capture in unmanaged and managed areas of the sites.

A replicated, controlled study in 2007–2009 on a plateau dominated by eucalypt forest in New South Wales, Australia (2) found that selectively removing trees on rock outcrops resulted in higher species richness of reptiles compared to overgrown outcrops with no tree removal, and increased abundance of four species, but decreased abundance for two. Species richness was higher on outcrops where trees were removed (5 species/outcrop) compared to overgrown outcrops (2 species/outcrop), and similar to outcrops that were naturally sun-exposed (5 species/outcrop). In outcrops with trees removed, the relative abundance of four of five sun-tolerant species increased, and two of two shade-tolerant species decreased (see original paper for details). In 2007, trees were selectively removed manually from 25 overgrown rock outcrops. Additionally, 30 overgrown (shady) outcrops and 20 naturally sun-exposed plots were selected that had no trees removed. Outcrops (of around 100 m^2) were separated by an average of 80 m. Reptiles were sampled monthly from May 2007–October 2009, and captured reptiles were marked.

A replicated, site comparison study (year not provided) in two sites of temperate broadleaf woodland on the border of Northamptonshire and Cambridgeshire, UK (3) found that more slow worms *Anguis fragilis* and common lizards *Zootoca vivipara* were found in woodland clearings maintained by cutting compared to in coppiced areas of a woodland. A total of 41 common lizards and 102 slow worms were found in clearings maintained by cutting, whereas no slow worms or common lizards were found in either recently coppiced sites (2–6 years previously) or older coppiced sites (9–17 years old). In each of two areas of woodland, three clearings maintained by vegetation cutting (one of the open areas was selected two weeks after surveys began), three sites of recently coppiced woodland (2–6 years old) and three sites of older coppice (9–17 years old) were selected. All coppiced areas were dominated by small-leaved lime trees *Tilia cordata*. At each survey site, 20 coverboards (50 x 50 cm; 10 made of roofing felt, 10 made of corrugated bitumen) were arranged in a grid, with 5 m gaps between boards. Coverboards were left for one week, and then checked for reptiles on 3–6 days/week for eight weeks.

A replicated, controlled, before-and-after study in 2006–2012 in mixed forest near Le Mans, France (4) found that after cutting trees to open the canopy, Asper vipers *Vipera aspis* were more likely to occupy areas with open than closed canopy habitats. Opening up the forest canopy caused an increase in the likelihood of viper occupancy (78%) compared to beforehand (34%). Overall, open forest was more likely to be occupied by snakes regardless of management (recently cut open canopy: 78%, open canopy maintained: 78%, open canopy cut 5–7 years ago: 69%) than closed canopy, unmanaged forest (9%). In winter 2006–2008, canopy cover was opened on four transects running alongside pre-existing public paths (5–10 m wide x 3,700 m long/transect by end 2008). In 2008–2012, transect segments were managed by: shrubs maintained at <2 m high (1,665 m total length); shrubs not managed (555 m); opening up more canopy (925 m); and unmanaged, mature forest with no historical cutting (555 m). Snakes were monitored using cover boards placed every 10–50 m along transect segments (76–202 boards/year). Boards were checked in April–September in 2006–2012 (23–86 survey days/year).

A replicated, controlled study in 2008–2012 in swamp forest and shrubland in New York State, USA (5) found that where canopy cover and shrubs were reduced, densities of eastern Massassauga rattlesnakes *Sistrurus catenatus* were higher in the first three years after cutting, but similar to uncut plots after four years. The effect of removing canopy or shrubs cannot be separated. Estimated rattlesnake densities were greater in 0-year-old (0.072–0.141 snakes/100 m^2), 1-year-old (0.045 snakes/100 m^2) and 3-year-old (0.133 snakes/100 m^2) cut plots than uncut plots (0.003–0.009 snakes/100 m^2). Rattlesnake densities in 4-year-old cut plots (0.013 snakes/100 m^2) were similar to uncut plots. Canopy was reduced by cutting shrubs to <0.25 m high in 50 plots in two known rattlesnake breeding areas in 2008 (six 28 m^2 plots), 2011 (thirty-two 100 m^2 plots) and 2012 (twelve 28 m^2 plots). In addition, 4 ha of adjacent forest was mechanically cleared in 2011. Snakes were monitored using visual encounter surveys in 66 plots with canopy removal (50 plots of cut vegetation in breeding areas and sixteen 36 m^2 plots within the forest areas cleared ion 2011), and 44 areas with no vegetation removal (twenty-eight 28 m^2 plots of uncut vegetation in breeding areas and sixteen 36 m^2 plots in uncut forest). In 2011 and 2012, the number of snakes caught in canopy removal areas (removal having occurred 0–4 years previously) was compared to the number of snakes in uncut plots. It is unclear whether the results reported are based on the breeding areas only or include the cut and uncut forest plots. Surveys were carried out once a week in June–August 2011 and May–August 2012. Snakes were captured, sexed and individually marked with PIT tags.

A before-and-after study in 2013–2014 in forest and wetland in Rhode Island, USA (6) found that clearing area patch of canopy did not affect spotted turtle *Clemmys guttata* home range size. In the year after a clearing was created by cutting trees, average spotted turtle home range size was similar to before the forest was cut (after cutting: 1.4 ha; before cutting: 1.2 ha). In December 2013–February 2014, a 3 ha area of mature forest was clearcut, leaving eight trees/ha and coarse woody debris on the ground and a 15 m border with adjacent wetlands. Twelve turtles were radio-tracked every five days in May–October 2013 (before clearcutting) and March–October 2014 (after clearcutting; 59 locations recorded/individual).

(1) Berglind S-Å (2005) Population dynamics and conservation of the sand lizard (*Lacerta agilis*) on the edge of its range. PhD Thesis. Uppsala University.
(2) Pike D.A., Webb J.K. & Shine R. (2011) Removing forest canopy cover restores a reptile assemblage. *Ecological Applications*, 21, 274–280.
(3) Fish A.C.M. (2015) Common lizards (*Zootoca vivipara*) and slow-worms (*Anguis fragilis*) are not found in coppiced Small-Leaved Lime (*Tilia cordata*) areas of a Northamptonshire-Cambridgeshire Nature Reserve. *Herpetological Bulletin*, 134, 26–27.
(4) Bonnet X., Lecq S., Lassay J.L., Ballouard J.M., Barbraud C., Souchet J., Mullin S.J. & Provost G. (2016) Forest management bolsters native snake populations in urban parks. *Biological Conservation*, 193, 1–8.
(5) Johnson B.D., Gibbs J.P., Bell T.A. & Shoemaker K.T. (2016) Manipulation of basking sites for endangered eastern massasauga rattlesnakes. *The Journal of Wildlife Management*, 80, 803–811.
(6) Buchanan S.W., Buffum B. & Karraker N.E. (2017) Responses of a spotted turtle (*Clemmys guttata*) population to creation of early-successional habitat. *Herpetological Conservation and Biology*, 12, 688–700.

Soil management

13.8. Disturb soil/sediment surface

- **Two studies** evaluated the effects of disturbing the soil/sediment surface on reptile populations. One study was in Sweden[1] and the other was in the USA[2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Abundance (1 study):** One replicated, controlled, before-and-after study in Sweden[1] found that after sand patches were created by soil scarification within clearings created by tree felling, sand lizards colonized, abundance then declined, but then increased once more, larger clearings were created[1].

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One randomized study in the USA[2] found that tilled areas were used more frequently by Blanding's turtles for nesting than mown or weeded areas.

Background
Many habitats depend on disturbances such as grazing to maintain structural habitat diversity, for example through trampling, and therefore support populations of species that depend on that disturbance. Tilling or soil scarification may be used as a way to disturb soils where grazing is not a viable option.

A replicated, controlled, before-and-after study in 1988–2004 in sandy pine heath in south-central Sweden (1) found where sand patches were created through disturbing the soil surface in areas where trees were cleared, sand lizards *Lacerta agilis* colonized, abundance then declined, but then increased after more sand patches in further clearings were created. After soil scarification and tree felling, sand lizards gradually colonized newly created habitat patches and eventually abandoned unmanaged habitat after 16 years (see original paper). During the first 10 years after sand patch creation, female sand lizard abundance declined (9–11%) but increased annually (11–19%) after a second, larger-scale clearing and sand patch creation programme was carried out. In particular, subadult lizard abundance increased more after the second creation programme (after first programme <10% increase; after second programme >150% increase in relative population size). Restoration was first carried out in 1988 and 1992 when nine 1–2 ha lizard-appropriate habitat patches in two sites were managed by tree felling and creating sand patches. A second restoration programme took place in the same two sites in 1999 and 2001, creating eighteen 10 ha habitat patches by tree felling, soil scarification and excavating 7–11 100–200 m^2 sand patches/site. Sand lizards were monitored from May–September in 1988–2004 by hand capture in unmanaged and managed areas of the sites.

A randomized study in 2006 and 2008 in wetlands in New York, USA (2) found that female Blanding's turtles *Emydoidea blandingii* preferred nesting in plots where soil was disturbed by tilling compared to weeded or mown plots. In 2006, nine of 10 monitored female turtles nested, of which seven nested in tilled plots and two in mowed plots. In 2008, six turtles nested, of which four nested in tilled plots, one in a weeded plot and one off the treatment plots. Two turtles nested in the same physical plot each year, in spite of a change in management. In 2006, thread trailing revealed that all female turtles explored or had been placed on each plot type before choosing where to nest. Eight sites around the edge of a fenced 12 ha wetland were monitored for turtle nesting activity. Each site contained three plots (5 x 7 m) with one of three managements randomly applied in 2006 and 2008: tilled to a depth of 15 cm (24 total plots), mowed to 5 cm height or 90% hand-weeded. Nesting activity was monitored by visual searches and radio tracking or by attaching a bobbin and thread to female turtles in May and June 2006 and 2008 (10 total turtles monitored).

(1) Berglind S-Å (2005) Population dynamics and conservation of the sand lizard (*Lacerta agilis*) on the edge of its range. PhD Thesis. Upsala University.
(2) Dowling Z., Hartwig T., Kiviat E. & Keesing F. (2010) Experimental management of nesting habitat for the Blanding's turtle (*Emydoidea blandingii*). *Ecological Restoration*, 28, 154–159.

Create habitat features

13.9. Add woody debris to landscapes

- **Six studies** evaluated the effects of adding woody debris to landscapes on reptile populations. Three studies were in Australia[4-6], two were in the USA[1,3] and one was in Indonesia[2].

COMMUNITY RESPONSE (3 STUDIES)

- **Richness/diversity (5 studies):** Four of five studies (including four replicated, randomized, controlled studies) in the USA[1,3], Indonesia[2] and Australia[5,6] found that areas with added woody debris had similar richness and diversity[1] or richness or of reptiles[2], rare reptiles[6] and snakes and lizards[3] compared to areas with no added debris. The other study[5] found that areas with added woody debris had higher reptile species richness than areas with no added debris.

POPULATION RESPONSE (6 STUDIES)

- **Abundance (6 studies):** Two of six replicated studies (including four randomized, controlled studies) in Australia[4-6], Indonesia[2] and the USA[1,3] found that areas with added woody debris had a higher abundance of reptiles than areas with no added debris[4,5]. Three studies[1-3] found that areas with woody debris had a similar abundance of reptiles[1,2] and snakes and lizards[3] compared to areas with no added debris. The other study[6] found that pastures with added timber had lower abundance of rare reptile species compared to pastures without timber, but that in pastures with added timber, reptile abundance was higher after 15 months than after 12 months.

BEHAVIOUR (0 STUDIES)

Background

Reptile species use both fine and coarse woody debris (sometimes called downed wood) as shelter habitat. In landscapes where debris has been removed, adding debris back may improve conservation outcomes (Bunnell & Houde 2010).

For studies discussing leaving woody debris and snags in place after logging or wood harvesting, see *Threat: Biological resource use – Leave standing/deadwood snags in forests* and *Leave woody debris in forests after logging*. For other studies that discuss providing shelter habitat, see *Create artificial refuges, hibernacula and aestivation sites*, and *Create artificial burrows*.

Bunnell F.L. & Houde I. (2010) Down wood and biodiversity—implications to forest practices. *Environmental Reviews*, *18*, 397–421.

A replicated, randomized, controlled study in 2002–2005 of pine stands in South Carolina, USA (1, same experimental set-up as 3) found that adding coarse woody debris (downed or standing) did not increase reptile abundance, species richness or diversity compared to areas with no debris added. Plots with added woody debris (downed and standing) were similar to unmanipulated plots in terms of reptile abundance (debris added: 0.3–0.5 individuals/plot vs unmanipulated: 0.4), richness (5–7 species vs. 7) and diversity (10–17 vs. x 13, Shannon-Weiner diversity index). Reptile richness was higher in plots with added downed debris (7 species) compared to plots with added standing debris (5 species). In 2002–2005, nine ha plots within three forest blocks had either downed woody debris added (3 plots, increased five-fold); standing woody debris added (3 plots, increased 10-fold) or were left unmanaged (3 plots). In 2002–2005, fourteen days of sampling were carried out each season (except in spring 2004, when there were 28 days) using drift fences with pit-fall traps.

A replicated, randomized, controlled, before-and-after study in 2007–2008 in cacao plantations Sulawesi, Indonesia (2) found that after woody debris, or debris and leaf litter, were added to plantations, both reptile abundance and species richness did not increase compared to in areas where no debris and/or leaf litter was added. All results were reported as statistical model outputs. Overall reptile abundance remained similar after woody debris or debris and leaf litter was added, but decreased after leaf litter and woody debris were removed, or when only woody debris was removed (see original paper for details of individual species abundance changes). Reptile species richness also remained similar after the addition of woody debris or debris and leaf litter, but decreased after leaf litter and woody debris were removed. Six plots (40 x 40 m^2) each in cacao plantations (number not specified) were randomly treated with: addition or removal of woody debris (trunks and branch piles), addition or removal of woody debris plus leaf litter or no management. Plots were sampled 26 days before and 26 days after habitat manipulation, three times a day in December 2007–July 2008. Active visual surveys were undertaken for 25 minutes along both plot diagonals (transects 3 x 113 m).

A replicated, randomized, controlled study in 1996–2008 in a loblolly pine *Pinus taeda* forest in South Carolina, USA (3, same experimental set-up as 1) found that increasing downed coarse woody debris had no effect on lizard or snake abundance, species richness or diversity compared to not manipulating debris. After adding debris, snake abundance, richness and diversity were similar (abundance: 0.03 individuals/m drift fencing, species richness: 0.02 species/m drift fencing, diversity: 0.003 Shannon-Wiener Index), to unmanipulated plots (0.04, 0.04, 0.01), but less than in plots with debris removed (0.07, 0.04, 0.01). For lizards there was no difference between adding (abundance: 0.15 individuals/m drift fence, species richness: 0.07 species/m drift fence, diversity: 0.02 Shannon-Wiener Index), not managing (0.01, 0.07, 0.02) or removing debris (0.15, 0.07, 0.02). Nine ha plots in three pine stands (approximately 45 years old, three plots/stand) were managed by: increasing volume of downed woody debris five-fold by felling trees (initiated 2001, to 59 m^3/ha in 2007); no manipulation of woody debris (initiated 1996, 13 m^3/ha woody debris); removing all downed woody debris ≥10 cm diameter and ≥60 cm in length by hand (initiated 1996, to 0.24 m^3/ha in 2006). All plots were prescribed burned in 2004. Reptiles were sampled for 14 days/plot in each of seven seasons (January 2007–August 2008) using drift fences with pitfall traps.

A replicated, controlled study in 2007–2010 in two grassy woodland reserves near Canberra, Australia (4) found that adding coarse woody debris in clumps only, or dispersed and in clumps, increased reptile abundance over four years, although the effect size depended on vegetation density and grazing intensity. Adding coarse woody debris (20 tonne/ha clumped or 40 tonne/ha clumped and dispersed) increased overall reptile abundance in one site and overall reptile abundance and small skink abundance in another site compared to not adding debris (results reported as model outputs). The effect of adding coarse woody debris was greatest in open vegetation compared to mid- or high-density vegetation, particularly when vegetation was subject to high-intensity grazing by kangaroos *Macropus giganteus* (see paper for details). Reptiles were monitored in 96 plots (1 ha) in 24 sites across two nature reserves (4 plots/site). In October 2007, coarse woody debris was added to 1 ha plots as follows: 20 tonnes/ha evenly dispersed (24 plots), 20 tonnes/ha in clumps (24), 40 tonnes/ha clumped and dispersed (24), or no coarse woody debris (24). In December 2007, six sites were fenced to exclude kangaroos and grazing levels were classed as low (fenced: 0.4 kangaroos/ha) or high (unfenced: 2.1). Reptiles were surveyed at each site using 30-minute active searches from March to April in 2007–2010.

A replicated, randomized, controlled study in 2011–2012 in upland forest in Queensland, Australia (5) found that reptile captures and species richness tended to be higher one year after coarse woody debris was added to restoration plantings compared to no debris added, or debris removed. Results were not statistically tested. One year after coarse woody debris was added to restoration plantings, reptile captures and species richness tended to be highest in restoration plantings with added coarse woody debris (captures: 3.7–4.0 individuals/site; species richness: 2.0 reptiles/site), followed by restoration plantings without added coarse woody debris (1.5, 0.7), and lowest in remnant forest without management (0.8, 0.5) or remnant forest with coarse woody debris removed (0.3, 0.2). In November 2011–January 2012, five treatments were applied four times each in four sites (60 m x 40 m sites): restoration planting (native trees and shrubs) with added salvaged log piles; restoration planting with added fence post piles; restoration planting with no debris added; remnant forest with no debris added; and remnant forest with all woody debris removed. Restoration plantings were 0–7 years old when coarse woody debris were added. Reptiles were surveyed in either March or August 2012 and again in December 2012.

A replicated, paired study in 2013–2015 in 12 pastures adjacent to grassy woodland in New South Wales, Australia (6) found that pastures with timber added had lower rare reptile abundance and similar species richness compared to pastures without timber, although abundance did increase underneath the timber over time. Rare reptile species abundance was lower in pastures with timber added (0.4 individuals/paddock) compared to pastures without timber (0.7 individuals/paddock). Rare reptile richness was similar in pastures with (1.4 species/paddock) and without timber (1.9 species/paddock). Reptile counts were higher at 15 months after timber installation (3.5 individuals/paddock) than at 12 months (1.4 individuals/paddock). In January 2014–March 2015, reptiles were surveyed in 12 farms grazed by sheep *Ovis aries* or cattle *Bos Taurus* with paddocks directly adjacent to remnants of native open grassy woodland. On each farm, two 80 m transects were surveyed: grazed pasture, and grazed pasture with timber added (50 x 50 x 40 cm timber pieces laid at 0.5 m intervals from the edge to 80 m into the pasture 2 months before the first surveys). Surveys were carried out using drift fences, pitfall traps and funnel traps set at 20, 50 and 80 m intervals/transect. Surveys took place for 5 days at a time in austral spring–summer. Rare species were defined as those captured in ≤4 sites with <70 total captures. Timber was checked for reptiles at 12 and 15 months after installation.

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13.10. Create artificial refuges, hibernacula and aestivation sites

- **Eleven studies** evaluated the effects of creating artificial refuges, hibernacula and aestivation sites on reptile populations. Three studies were in each of the UK[2,4,9] and Australia[3,6,11], two were in New Zealand[5,8] and one was in each of the USA[1], Spain[7] and Italy[10].

COMMUNITY RESPONSE (1 STUDY)

- **Richness/diversity (1 study):** One controlled, before-and-after study in Spain[7] found that areas with refuge logs had higher reptile species richness than areas without refuges.

POPULATION RESPONSE (3 STUDIES)

- **Abundance (1 study):** One controlled, before-and-after study in Spain[7] found that areas with refuge logs had a higher abundance of reptiles than areas without refuges.
- **Reproductive success (1 study):** One study in the UK[9] found that after translocating adders to an artificial hibernaculum, there was evidence of successful reproduction.
- **Survival (1 study):** One randomized, controlled, before-and-after study in New Zealand[8] found that in areas with artificial refuges, survival of McCann's skinks was similar to areas without refuges.

BEHAVIOUR (9 STUDIES)

- **Use (9 studies):** Nine studies (including one replicated, controlled study and one randomized, controlled study) in the USA[1], the UK[2,4,9], Australia[3,6,11], New Zealand[5] and Italy[10] found that artificial refuges were used by reptiles[1,3,9], common lizards[2,4], adders[2,9], common geckos[5], species of skinks[5,6,11], and by an ocellated lizard to lay a clutch of eggs[10]. Four of the studies[3,5,6,11] also found that some reptiles showed a preference for refuges with certain designs or construction materials.

Background

Reptiles need shelter for overwintering or aestivating during hot arid summers. Artificial overwintering or aestivating sites, or 'hibernacula', can be created for reptiles where natural sites are limited or where these habitats have been lost, for example at newly restored sites or in gardens.

Other studies investigating the creation of shelter habitat are discussed in *Add woody debris to landscapes*; *Create artificial burrows*; *Create or restore rock outcrops*; *Threat: Biological resource use – Leave woody debris in forests after logging* and *Leave standing/deadwood snags in forests*.

A replicated, controlled study in 1977–1979 in three riverine forest sites in Louisiana and Mississippi, USA (1) found that artificial nest boxes were used by six reptile species. In total six reptile species were found in nest boxes and occurred in 0.3–11.3% of large boxes, 0.4–5.9% of medium boxes and 1.3–5.4% of small boxes compared to 1.2–2.0% of natural tree cavities (reptile numbers and species not provided). Boxes were erected in hardwood and hardwood/pine forests and were of three sizes: large (60 x 30 x 30 cm, 13 cm diameter entrance), medium (45 x 20 x 20 cm, 7.5 cm diameter entrance) and small (30 x 15 x 15 cm, 5 x 7 rectangle entrance). Fifty boxes were installed at two sites and 90 at the other. All boxes had 5–10 cm of pine shavings in the bottom. Boxes and natural cavities were inspected every month from April 1977 to February 1979.

A study in 1999 on a heathland site in Berkshire, southern England, UK (2) found that an artificial hibernaculum was used by common lizards *Zootoca vivipara* and adders *Vipera berus*. Following construction, three adult lizards were observed basking near entrance holes and three adder skins were discovered. An artificial hibernaculum was constructed 40 m away from a bank that was to be destroyed as part of a road development. A ditch was dug (20 x 1 x 1 m) and hollow concrete building blocks were used to create underground chambers, with plastic piping (5 cm diameter) providing entrance tunnels. Bark mulch was added to any gaps and the structure was backfilled and covered with turf and native shrubs. Observations of reptiles at the hibernaculum were conducted on one day in April 1999.

A replicated, site comparison study in 2000–2001 in a site of grassland with wooded patches in Victoria, Australia (3) found that more reptiles tended to use old log refuges compared with new log refuges. Three species were found more commonly under old logs compared to new logs (tessellated gecko *Diplodactylus tessellatus*: 6 individuals in old logs vs 2 in new logs, Boulenger's skink *Morethia boulengeri*: 12 vs 6; curl snake *Suta suta*: 38 vs 7). Three species were found in similar numbers under old and new logs (striped legless lizard *Delma impar*: 1 in old logs vs 2 in new; olive legless lizard *Delma inornate*: 6 vs 9; Grey's skink *Menetia greyii*: 23 vs 9) and two species were found under only one log type (bearded dragon *Pogona barbata*: 1 under new log; eastern brown snake *Pseudonaja textilis*: 2 under old log). An area of 3,780 ha was marked into 91 quadrats and in May 2000, and 12–20 logs (old fence posts) were placed in every quadrat (total of 1,131 log refuges). An additional 271 fallen fence posts that had lain in situ for 15 years were also monitored. Monthly surveys took place between June 2000 and January 2001.

A study in 2004–2005 in scrub and grassland in Suffolk, UK (4) found that artificial hibernacula were used by translocated common lizards *Lacerta zootoca vivipara*. Six months after lizards were first translocated to the hibernacula, both adult and juvenile lizards were observed basking around each hibernaculum. Three hibernacula were constructed (east-west ditches 20 m long, 1 m deep and 1.5 m wide with approximately 70° sloping edges) and filled with a mixture of drainage pipes, bricks, gravel, rubble, vegetation cuttings, logs and soil in autumn 2004. Plastic piping was added to facilitate lizards entering and entrances restricted in size to limit access by predators such as weasels *Mustela nivalis* and brown rats *Rattus norvegicus* (see original paper for details). Approximately 70 lizards were caught and translocated in autumn 2004 and spring 2005. Lizard use of the hibernacula was monitored from March 2005.

A replicated study in 2003–2004 in two grazed farmland sites near Canterbury, New Zealand (5) found that artificial refuge design was important for common geckos *Hoplodactylus maculatus* but not McCann's skinks *Oligosoma maccanni* or common skinks *Oligosoma nigriplantare polychroma.* Common geckos preferred artificial refuges made from Onduline (a corrugated roofing product made of organic fibers: 602 total captures) compared to corrugated iron (109 total captures) or concrete roofing tiles (27 total captures). Similar numbers of McCann's skinks and common skinks were captured under each artificial refuge material (McCann's Onduline: 28 total captures vs. iron: 22 vs. concrete: 36; common skink 21 vs. 23 vs. 30). The refuges were triple-layered and common geckos were captured 344 times in the top spaces, 316 times in the middle spaces and 51 times in the bottom spaces. At each site, a 5 x 6 grid of 'refuge stations' spaced 5 m apart was installed. Each station consisted of three triple-layered artificial refuges made of different materials: Onduline, iron and concrete roofing tiles. All refuges were checked monthly from December 2003 until November 2004.

A replicated study in 2004–2005 in fenced sand and grass enclosures in South Australia, Australia (6) found that gidgee skinks *Egernia stokesii zellingi* preferred artificial refuge structures with more crevices than those with fewer. Skinks spent more time on artificial refuge structures with more crevices (41 minutes/skink) than on those with fewer crevices (16 minutes/skink). Skinks spent more time taking refuge in the crevices of artificial refuges with more crevices (25 minutes/skink) than in those with fewer crevices (5 minutes/skink). Artificial refugia were created from 3 cm thick concrete slabs (40 x 40 cm or 60 x 60 cm) and placed in four outside pens (3 x 1.4 m) with a sand and grass substrate. For each trial, two refugia were provided at each end of the pen (60 cm apart). Each refuge had a base (1.2 x 1.2 m) made of four slabs. One, four or eight crevices were added to each structure using timber or slabs (see original paper for details). Skinks used in the trials were from a captive colony. Skinks were individually marked with paint prior to being placed in a pen (1 or 4 individuals at a time) and left undisturbed for 20 minutes. Skink behaviour was monitored by video camera for the following 60 minutes. Thirty trials were carried out in September 2004–March 2005.

A controlled, before-and-after study in 2000–2006 of a riparian site of Mediterranean shrubs in southwestern Spain (7) found that restoration sites with refuge logs had higher abundance and species richness of reptiles than sites without logs. After 2–4 years, the site with refuges hosted more reptiles than the site with no refuges (refuges: 4–7 individuals/hour; no refuges: 1–3 individuals/hour) and the number of species seen/hour was also higher (refuges: 1.4–1.7 species/hour; no refuges: 0.8–1.3 species/hour). Overall species richness after 2–4 years was similar for the site with refuges (6 species) and a nearby intact site (7 species), and lower for the site with no refuges (5 species) compared to the intact site. Large scale restoration of a riparian corridor (4,200 ha) began following a mining accident in 1998. In 2002, one 24 ha site was provided with 120 reptile refuges: two logs (1.2 m long) placed side by, distributed evenly across the site. Another site (24 ha) received no logs. An additional site outside the affected corridor was also sampled. Reptile surveys began in 2000, and in 2002–2006, at least three surveys were carried out each year, each lasting 4–5 hours.

A randomized, controlled, before-and-after study in 2004–2006 on a coastal duneland site on South Island, New Zealand (8) found that providing artificial refuges for McCann's skink *Oligosoma maccanni* did not lead to an increase in survival compared to when no refuges were provided. Average change in skink survival before and after refuges were provided did not differ from zero. Change in survival was also no different from zero when artificial refuges and exclosure fencing were provided together. Four sites each were assigned to one of four treatments: artificial refuges (32 refuges/site, 16 each of two designs); artificial refuges and exclosure fences (25 x 25 m area, 1 m high chicken wire fence, bird netting on top); exclosure fences only; and no treatment. Refuges were made of corrugated roofing and cladding. Skinks were sampled annually using a 4-day pitfall trapping session in February and March 2004–2006 with fencing and refuges placed into randomly allocated sites immediately before the second year.

A study in 2009–2011 in grazed marsh in Norfolk, UK (9) found that some translocated adders *Vipera berus* released onto man-made hibernacula bred, returned to the hibernacula to overwinter and survived for at least 18 months. Six months after translocation, up to 22 adders/day were recorded on the man-made hibernacula, including one newborn adder, indicating breeding success. Eighteen months after translocation, 21 of 119 translocated adders were sighted on or near the hibernacula. In addition, 19 new adders were observed in the vicinity. Viviparous lizards (including juveniles) and grass snakes *Natrix helvetica* were also recorded on and near the hibernacula 12–18 months after they were built. In September 2009, three hibernacula (100 m approximate length; 1.5 m high, 3 m wide with 45° front and rear slopes) were constructed from natural materials on grazing marshes separated by drainage ditches. Each hibernacula and some adjacent grazed land (1 ha total) were enclosed by semi-permanent fencing (plastic sheeting and wooden posts). In March 2010, a total of 119 adders were translocated from nearby flood banks that were subject to flood defence works (which took place May-October 2010). The fencing was opened from mid-May 2010. Adders were monitored in September–October 2010, March–May and July–September 2011.

A replicated study in 2013–2015 in an area of Mediterranean shrubland in Savona Province, Italy (10) found that one of six artificial shelters consisting of a concrete block was used by a female ocellated lizard *Timon Lepidus* to lay a clutch of eggs. Two years after six artificial shelters were installed, a single female lizard laid a clutch of nine eggs in one of the shelters. Two months later the shelter was found to have been destroyed and the fate of the eggs was unknown. In 2013, six artificial shelters were installed that consisted of a hollow concrete brick (12 x 7 cm opening and 40 cm deep) camouflaged by stones and branches. Shelters were monitored in March–October: seven times in 2013, once in 2014 and twice in 2015.

A randomized, controlled study (years not provided) of artificial refugia in Australia (11) found that Boulenger's skinks *Morethia boulengeri* preferred timber refuge material compared to cement tiles or corrugated iron, but that this preference was affected by the size of the gap between the refuge and the ground. Skinks selected timber refuges over corrugated iron refuges (timber: 21 skinks; iron: 6 skinks) and timber over cement tiles (timber: 19 skinks; cement: 9 skinks), but showed no preference for corrugated iron or cement tiles (iron: 14 skinks; cement: 14 skinks). When the preferred timber refuges were raised from 1 cm to 2.5 cm above ground, all skinks (10 of 10) preferred corrugated iron with gaps of 2 cm, but preference for standard timber (2.5 cm gaps) and flattened iron (<=1 cm gaps) was equal (5 skinks selected each). Twenty-eight skinks collected from two different areas were presented with choices between two different refuge materials (either timber, corrugated iron or cement tile). Twenty of those skinks were then given the choice of a timber refuge or corrugated iron refuge raised to different heights above the ground (timber: height changed from 1 cm to 2.5 cm above ground; corrugated iron was flattened from 2 cm gaps to <=1 cm). Experiments were carried out in laboratory conditions.

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13.11. Create artificial burrows

- **Six studies** evaluated the effects of creating artificial burrows on reptile populations. Five studies were in Australia[2-6] and one was in the USA[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Abundance (1 study):** One controlled, before-and-after study in Australia[3] found that areas with artificial burrows had more pygmy blue tongue lizards than areas with no artificial burrows
- **Reproductive success (1 study):** One replicated, controlled study in Australia[2] found that female pygmy bluetongue lizards using artificial burrows produced larger offspring than those using natural burrows.
- **Condition (1 study):** One replicated, controlled study in Australia[2] found that female pygmy bluetongue lizards using artificial burrows had better body condition than those using natural burrows.

BEHAVIOUR (5 STUDIES)

- **Use (4 studies):** Three replicated studies (including one controlled study) in Australia[2,5,6] found that artificial burrows were used by resident[2,6] and translocated[5] pygmy bluetongue lizards[2,6]. One of the studies[6] also found that pygmy bluetongue lizards preferred artificial burrows with a chamber than burrows with no chamber. One replicated study in the USA[1] found that providing artificial burrows for translocated gopher tortoises resulted in more tortoises settling successfully in the release area.
- **Behaviour change (1 study):** One replicated, controlled, before-and-after study in Australia[4] found that translocated pygmy blue tongue lizards used artificial burrows, and supplementary food affected the amount of time they spend in bare ground areas.

Background

Artificial burrows may be created for reptiles to support new or existing populations where natural burrows are limited or have been lost, for example at newly restored sites.

Other studies investigating the creation of shelter habitat are discussed in *Create artificial refuges, hibernacula and aestivation sites*; *Add woody debris to landscapes*; and *Threat: Biological resource use – Leave woody debris in forests after logging* and *Leave standing/deadwood snags in forests*.

A replicated study in 1980–1982 in five areas of pine forest in Mississippi, USA (1) found that providing artificial burrows inside release pens when translocating gopher tortoises *Gopherus Polyphemus* tended to result in more successful translocations than releasing tortoises directly into the wild. Results were not statistically tested. When translocated gopher tortoises were released into artificial burrows within release pens before being released into the wild, 17 of 21 translocations were successful. Zero of three translocations were successful when tortoises were released into artificial burrows with no pen; one of five when released into a natural burrow with no pen; and zero of 11 when no burrow or pen was provided. Forty individually-marked adult gopher tortoises (some may have been captive releases) were translocated in spring–summer 1980–1982 (one tortoise = one translocation). Tortoises were released directly into artificial burrows in the wild (1 m deep; 3 tortoises); into artificial burrows in circular release pens (4–7 m diameter pens; 21 tortoises, pen removed after 2–4 weeks); into abandoned natural burrows in the wild (5 tortoises); or were released directly into the wild with no specific management (11 tortoises). Tortoises were monitored until late summer or early autumn in the release year and translocations were judged successful if after release in to the wild, previously abandoned burrows became active and a translocated tortoise was found in them, or new tortoise burrows were dug in areas without pre-existing tortoise populations.

A replicated, controlled study in 1995–1998 in a grassland site in South Australia, Australia (2) found that female pygmy bluetongue lizards *Tiliqua adelaidensis* using artificial burrows had better body condition and produced larger offspring than female lizards using natural burrows. Females observed in artificial burrows had a higher body condition index than those in natural burrows (data reported as statistical model result). Females in artificial burrows also produced heavier offspring (artificial: 1.7 g; natural: 1.5 g) with a higher body condition index (artificial: 2.7; natural: 2.6) than females in natural burrows, though snout-vent length of offspring and females was similar for both groups (offspring: artificial: 44.5 mm; natural: 44.4 mm; females: artificial: 97.9 mm; natural: 96.5 mm). Body condition of males was similar in artificial and natural burrows. One-hundred artificial burrows (10 m apart in 10 x 10 grid, 30 cm deep and 1.7 cm in diameter) were added in August 1995 to a 1 ha plot adjacent to a natural population. They were made by hammering a metal rod into the ground and inserting a hollow wooden tube. Burrows were monitored weekly using a fibre optic camera from September to May over a three-year period. Lizards were lured out of natural burrows (147 females, 124 males) or removed in the tube from artificial burrows (40 females, 49 males).

A controlled, before-and-after study in 2000–2002 in grassland in South Australia, Australia (3) found that the addition of artificial burrows resulted in an increase in the number of pygmy blue tongue lizards *Tiliqua adelaidensis*. The average number of lizards in plots with artificial burrows increased following installation of burrows (before: 1 lizard/plot; 4 months after: 4 lizards/plot; 7 months after: 7 lizards/plot), while numbers on the plots without artificial burrows remained stable through the three surveys (1.4; 1.4 and 1.5 lizards/plot). In April 2002 (after new juveniles have left birth burrow), plots with artificial burrows had more juveniles (3.5 juveniles/plot) than those with natural burrows only (0.9 juveniles/plot). The average number of lizards in natural burrows did not change significantly with year or treatment (0.8–1.5 lizards/plot). The experiment was conducted in a 300 x 140 m area adjacent to a 1 h monitoring area. Twenty-four 20 x 20 m plots were established with an average of 3–5 natural burrows of 12 cm or deeper. After an initial survey in August 2001, eighteen small (13 mm diameter, 30 cm deep) and 18 large (17 mm diameter, 30 cm deep) artificial burrows were added to 12 experimental plots. Burrows were monitored using an optical fiber scope.

A replicated, controlled, before-and-after study in 2009 in grass, bare ground and tilled soil enclosures in southern Australia (4, same experimental set-up as 5) found that translocated pygmy bluetongue lizards *Tiliqua adelaidensis* used artificial burrows, and lizards given supplementary food spent less time in open habitat away from the burrows. Of 2,298 recorded lizard behaviours, 1,352 were of lizards basking in burrow entrances before re-entering; 708 were of lizards fully emerging and returning to the same burrow; and 238 were of lizards emerging and entering a different burrow. Lizards provided with supplementary food spent less time out in open habitat than lizards that were not fed (see paper for details). In November 2009, sixteen lizards were captured and moved to a trial site in a zoo and placed in four 15 m enclosed cages (four lizards/cage). Cages contained short grass, bare ground and tilled soil. Artificial burrows were built from hollowed wooden poles (30 cm long, 3 cm diameter) pushed into grassy or tilled soil (82 burrows/cage). No burrows were present in the bare ground habitat. Lizards in two of the cages were provided supplementary food for seven days, then after a two-day break, lizards in the other two cages were provided supplementary food for seven days. Lizards were monitored by four surveillance cameras/cage during daylight hours from the second to seventh days of the feeding regime (12 days total).

A replicated study in 2009 in a grassy enclosure in South Australia, Australia (5, same experimental set-up as 4) found that translocated pygmy bluetongue lizards *Tiliqua adelaidensis* used artificial burrows, and burrow use was similar whether lizards were confined to holding pen for one or five days prior to release. Lizards were observed basking at artificial burrow entrances 85% of the time and exiting burrows 14% of the time. Of movements to and from artificial burrows, 62% were lizards returning to the same burrow, 29% were lizards moving to new burrows in the centre of the enclosure and 9% were lizards moving to new burrows at the edge of the enclosure. Lizard movements between artificial burrows was similar between translocated lizards confined to a holding area with burrows for one or five days (data reported as model outputs). In October 2009, sixteen translocated pygmy bluetongue lizards were released into one of four cages in a zoo enclosure (4 lizards/cage). Each cage included a central grassy circle (4 m diameter) with artificial burrows (made from hollowed wooden rods pushed into the ground), surrounded by a strip of bare ground (5 m wide), encircled by a strip of marginal habitat (0.5 m wide) with artificial burrows. When lizards were released, all cages had a holding pen around the central grass areas. The pen was removed after one (two cages) or five days (two cages). Lizard activity was monitored by video cameras over 10 days and analysis of lizard behaviour was based on observations from days 6–10 of the study (capturing 3,535 activity events and 504 lizard movements).

A replicated study in 2011 in laboratory conditions in South Australia, Australia (6) found that pygmy bluetongue lizards *Tiliqua adelaidensis* preferred to use artificial burrows with a chamber at the end, regardless of the size of the chamber. (All results were presented as model outputs unless otherwise stated). Lizards spent more time in artificial burrows with chambers attached (196–354 minutes in burrows with tennis-ball chambers) compared to burrows without chambers (6–97 minutes in burrows without chambers). Lizards spent similar amounts of time in artificial burrows with large and small chambers and also did not show any preference for whether the burrow was lined with sand or not (data reported as statistical model result). In June–July 2011, five different artificial burrows were tested (all surfaces lined with glued-on sand unless otherwise stated): plastic tube without a chamber; plastic tube with tennis ball chamber; plastic tube with large chamber; plastic tube with small chamber; and a plastic tube with large container and no sand lining (see paper for tube and container dimensions). Eight of 12 wild-caught lizards were used in each trial (3–4 trials/lizard). Lizard responses were videoed for 6 hours/day over two days.

(1) Lohoefener R. & Lohmeier L. (1986) Experiments with gopher tortoise (*Gopherus polyphemus*) relocation in southern Mississippi. *Herpetological Review*, 17, 37–40.
(2) Milne T., Bull C.M. & Hutchinson M.N. (2003) Fitness of the endangered pygmy blue tongue lizard *Tiliqua adelaidensis* in artificial burrows. *Journal of Herpetology*, 37, 762–766.
(3) Souter N.J., Bull C.M. & Hutchinson M.N. (2004) Adding burrows to enhance a population of the endangered pygmy blue tongue lizard, *Tiliqua adelaidensis*. *Biological Conservation,* 116, 403–408.
(4) Ebrahimi M. & Bull C.M. (2012) Food supplementation reduces post-release dispersal during simulated translocation of the Endangered pygmy bluetongue lizard *Tiliqua adelaidensis*. *Endangered Species Research*, 18, 169–178.
(5) Ebrahimi M. & Bull C.M. (2013) Determining the success of varying short-term confinement time during simulated translocations of the endangered pygmy bluetongue lizard (*Tiliqua adelaidensis*). *Amphibia-Reptilia*, 34, 31–39.
(6) Staugas E.J., Fenner A.L., Ebrahimi M. & Bull C.M. (2013) Artificial burrows with basal chambers are preferred by pygmy bluetongue lizards, *Tiliqua adelaidensis*. *Amphibia-Reptilia*, 34, 114–118.

13.12. Create artificial nests or nesting sites

- **Nine studies** evaluated the effects of creating artificial nests or nesting sites on reptile populations. Three studies were in the USA[6-8] and one study was in each of the Galápagos[1], Spain[2], China[3], Reunion Island[4], Canada[5] and Jamaica[9].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (6 STUDIES)

- **Reproductive success (6 studies):** Two studies (including one before-and-after study) on Reunion Island[4] and Jamaica[9] found that the number of Reunion day gecko eggs[4] and Jamaican iguana hatchlings[9] at artificial nesting sites increased over time. One of two replicated, controlled studies in Canada[5] and the USA[6] found that hatching success of eggs from four species of freshwater turtle[5] moved to artificial nest sites was higher than for eggs left in natural sites. The other study[6] found that hatching success of diamondback terrapin nests in artificial nest sites compared to natural sites varied depending on the substrate used. One study in Spain[2] found that eggs laid in an artificial nest by an Iberian wall lizard hatched and those placed in artificial nests had high hatching success. One replicated study in the USA[7] found that fewer diamondback terrapin nests were predated in artificial nesting mounds protected with an electric wire than in mounds with no wire.

BEHAVIOUR (8 STUDIES)

- **Use (8 studies):** Four of seven studies (including one replicated, controlled study) in the Galápagos[1], Spain[2], Reunion Island[4], Canada[5], the USA[7,8] and Jamaica[9] found that artificial nest sites were used by captive Galápagos giant tortoises[1], Iberian wall lizards[2], four species of freshwater turtle[5] and diamondback terrapins[7]. Two studies[4,9] found that use of artificial nest sites increased over time for Reunion day geckos[4] and Jamaican iguanas[9]. The other study found that artificial nest sites were used infrequently by northern map turtles[8]. One study in China[3] found that artificial nesting materials were used by some Chinese alligators.

Background

Providing artificial nesting sites or nests may give reptiles the opportunity to lay eggs in appropriate settings where the natural environment is lacking.

For studies discussing the provision of artificial shade due to high temperatures for nesting sites or nests, see *Threat: Climate change and severe weather – Provide artificial shade for nests or nesting sites*.

A replicated, before-and-after study in 1965–1971 in a captive breeding facility in the Galápagos, Ecuador (1) found that Galápagos giant tortoise *Geochelone elephantopus hoodensis* females successfully laid eggs when artificial nesting sites mimicking natural conditions were provided. Results were not statistically compared. During the first two nesting seasons, when no artificial nests were provided, females attempted to nest on successive evenings (approximately 20–30 nights attempted nesting/clutch) but eventually dropped eggs on the rocky surface (four clutches). After artificial nests without ideal substrate were provided, females attempted to nest on successive evenings (10–30 nights/clutch) and some eggs were deposited in artificial nesting sites (artificial sites: 2 nests; on rocky substrate: 2 nests). After provision of ideal soil substrate, nesting attempts were shorter (1–4 nights/clutch, rarely up to 12) and all eggs were deposited in artificial (17 nests) or natural sites (2 nests). One male and 10 female Galápagos giant tortoises were brought into a captive breeding enclosure to mate and nest from the 1967/1968 nesting season. In 1969/1970, three artificial nesting sites were built (coarse soil, minimum 3 m^2 and 35–40 cm deep). These were removed in the 1970/1971 and 1971/1972 nesting seasons, and replaced with four artificial nest sites with fine soil identical to that found in the natural nesting area (all other dimensions the same).

A study in 1993 on an island in the Columbretes archipelago, Spain (2) found that all of 15 artificial nests were visited by Iberian wall lizards *Podarcis hispanica atrata* and one nest contained a clutch of laid eggs. All 15 artificial nests were used as basking and burrowing sites by adult male and female Iberian wall lizards (rocky area: total 3–17 active lizards/nest; vegetated area: 7–35 active lizards/nest). One artificial nest (in the vegetated area) had a clutch of two eggs laid in it, which hatched successfully. A total of 39 of 47 introduced eggs (83%) survived and developed successfully in the artificial nests. Fifteen white plastic containers (20 x 15 x 7 cm) filled with volcanic sand (five with rocks, five with stone shingle) were placed in a rocky area with a low density of lizards (seven containers; <100 lizards/ha) and a vegetated area with a high density of lizards (eight containers; 800 lizards/ha). Containers were covered with stone slabs and placed on the ground 5–15 m apart surrounded by rocks. Water was added every other day. In May–July 1993, lizards and their faecal pellets and burrows were counted during 40 x 5-minute observations over 19–20 days. All 15 containers were searched for naturally laid eggs on two occasions. Three introduced eggs (laid by captive female lizards) were placed in each container and survival recorded weekly.

A study in 2009 in vegetated pond banks in Anhui, China (3) found that Chinese alligators *Alligator sinensis* nested at a quarter of sites where artificial nesting materials were provided. Chinese alligators constructed nests at 11 of 43 sites where artificial nesting materials were provided. In addition, Chinese alligators constructed nests at eight locations across the whole study area where nesting materials were not provided. In May 2009, artificial nesting materials were provided in 43 sites in an outdoor alligator captive-breeding enclosure. The enclosure was surrounded by a 2.1 m high fence and included eight natural ponds surrounded by native vegetation. In total, 211 adult Chinese alligators had been introduced to the enclosure (0.024 alligators/m^2). The banks of all ponds (areas with and without provided nesting material) were monitored daily in the first part of July 2009 for signs of nesting activity.

A study in 2009–2011 in tropical rainforest on Reunion Island, Indian Ocean (4) found that some artificial egg-laying sites in a habitat restoration area were used by Reunion day geckos *Phelsuma borbonica* in the year they were installed and the number of sites used and eggs laid increased in the second year. Nine months after artificial egg-laying sites were installed, four of 34 sites were used by geckos and 10 eggs were laid. Two years after the first artificial egg-laying sites were installed, eight of 40 sites were used by geckos and 41 eggs were laid. In total, 40 artificial egg-laying sites were added to an area (9,000 m^2) of degraded habitat in a hydroelectric power plant in September 2009–July 2010 (34 were installed by June 2010 and a further 6 by July 2010). Artificial egg-laying sites comprised hollow, rectangular metal poles (4 x 8 x 250 cm) inserted into the ground (50 cm deep). Native plant species were planted in the same area to restore habitat (22,000 plants of 50 species). Egg-laying sites were monitored for signs of geckos and egg laying in June and September 2010, and March and September 2011.

A replicated, controlled study in 2009–2010 in a mosaic of wetlands, rivers and lakes in Ontario, Canada (5) found that freshwater turtle species used artificial nest mounds more than expected, and eggs in artificial mounds had higher hatching success than eggs left in natural nests. Turtles used artificial nests more than expected by chance (artificial mounds constituted 2% of nesting habitat but hosted 4% of nests). Of the four turtles that used the artificial mounds (1 painted turtle *Chrysemys picta*, 1 snapping turtle *Chelydra serpentina*, 2 Blanding's turtle *Emydoidea blandingii*), all had 100% hatching success. Eggs transplanted to artificial nests had higher hatching success than those left in natural nests for nine painted turtle nests (artificial: 98%; natural 71%) and 12 snapping turtle nests (artificial: 88%; natural 56%). Four artificial nesting mounds (60% gravel and 40% sand) 6 m diameter and 0.5 high were installed in April 2009 on top of a layer of geotextile cloth. Each mound was within 100 m of water, 50 m of a known nesting site and sited to prevent nesting turtles from having to cross a road. All natural, artificial and potential nesting mounds within 1 km of each artificial mound were monitored nightly from May-June 2009–2010. For the transplant experiment, nests were excavated and split evenly between the closest artificial mound and the original nest. Hatching events were monitored from August, and nests were excavated in October to assess hatching success.

A replicated, controlled study in 2006–2007 on an island of salt marsh grasses in New Jersey, USA (6) found that hatching success of diamondback terrapin *Malaclemys terrapin* nests in artificial nesting mounds varied depending on the construction material and year when compared to natural nests. Dredge soil mounds produced no hatchlings in the first year (0 of 10 nests hatched) but had some success in the second year (10 of 12 nests hatched, 42–60% hatching success). Loamy-sand mounds produced hatchlings in both years (11–85% hatching success). Hatching success in sand mounds varied from 0–31% in the first year and 41–65% in the second year. Natural nests had hatching success of 54% in the first year and 70% in the second year. Three experimental plots (2.25 m^2) were filled with 45cm of soil: dredge soil from a nearby channel which had been dried for two months; loamy sand from a natural nesting area or sand from a beach. One half of each plot was shaded by shade cloth 15 cm above the soil with the other half in full sun and each nest had a predator excluder made of wire mesh. Natural nests were in full sun with nearby vegetation cover. Clutches were relocated to treatment plots from areas with high human activity (2006: 5 nests/treatment, 5 natural controls; 2007: 6 nests/treatment, 8 natural controls). Nests were excavated after 60 days to assess hatching success.

A replicated study in 2013–2014 on an island site between a saltmarsh and road in Georgia, USA (7) found that diamondback terrapins *Malaclemys terrapin* made use of artificial nest mounds, and an electrified nest box provided more protection from predation than a nest box alone. At least 37 nests were laid in nest mounds (number of confirmed nests from table), yielding at least 203 hatchlings. Fewer nests laid under a nest box with an electric wire were predated (1 of 27 nests found) compared to those under a nest box with no wire (16 of 16 nests found). An artificial nesting mound (22.9 m long × 3.6 m wide × 1.2 m tall) was constructed using dredge material along the shoulder of an 8.7 km causeway leading to the island. On top of the mound were placed six nest boxes (3.7 x 1.2 x 0.6 m) with a ground-level 9 cm horizontal gap to allow terrapins access but to exclude predators. For 35 days from May–June 2013, one nest box was modified to include a battery-powered electric wire along the horizontal gap opening and for 26 days from June–July 2013, all six nest boxes had electric wires. The mound was excavated to find nests and hatched eggs in November 2013 and April 2014.

A study in 2000–2008 on a roadside verge along a river bank in Pennsylvania, USA (8) found that sand and shale mounds built along a barrier fence as mitigation nesting habitat after a road was constructed were used by a small number of nesting female northern map turtles *Graptemys geographica* in the first year. In the first year following creation of sandy mounds as nesting sites, two of 50 nests were laid in the sand mounds. The authors reported that most females walked over the sand mounds and nested near the barrier fence. In 1999, a new highway was built along a major river and in 2000 a chain-link fence (1 m high, 1,150 m long) was erected to mitigate road deaths of female turtles crossing the road to find suitable nesting habitat. Eight mounds of sand (800 m^3 total volume) were created on the river side of the road fence to provide nesting habitat. In 2001, the sand was moved closer to the fence and shale was added to reduce vegetation. Turtle nesting was monitored in May–July 2000–2003, 2005–2007 and 2008 (dates not provided for 2008) in the mitigation nesting habitat and at another commonly used nesting site, but data on use of mitigation nesting sites were only provided for 2000.

A before-and-after study in 1991–2015 in old-growth dry limestone forest in Jamaica (9) found that when an artificial nesting site was created as part of a

Jamaican iguana *Cyclura collei* head-starting programme, numbers of nesting female and hatchling iguanas increased over 23 years. Results were not statistically tested. Twenty-three years after the start of a Jamaican iguana head-starting programme involving building an artificial nesting site, 321 iguana hatchlings and 63 nesting female iguanas were counted compared to 31 hatchlings and nine nesting females at the start of the programme. Two nests were laid in the artificial nest site three years after it was built. In 1991–2015, Jamaican iguana eggs/hatchlings were collected for head-starting in a zoo and head-starters were released from 1996 (278 total head-starters released, usually 6–8 years old or 1–2 kg). In 1997–2014, non-native mammalian predators (mongoose *Herpestes javanicus*, cats *Felis catus*, dogs *Canis lupus familiaris* and feral pigs *Sus* sp.) were removed using baited cage traps, snares and leg-hold traps (~1,500 individual removed in ~350,000 trap days over 17 years using 20–300 cage traps). In 2011–2012, an artificial nesting site was constructed 40 m south of the main nesting area. During the nesting season in 1991–2015, nests were checked daily and adult female iguanas were monitored by live trapping, observation and camera traps.

(1) MacFarland C.G., Villa J. & Toro B. (1974) The Galápagos giant tortoises (*Geochelone elephantopus*) Part II: Conservation methods. *Biological Conservation*, 6, 198–212.

(2) Castilla A.M. & Swallow J.G. (1995) Artificial egg-laying sites for lizards: A conservation strategy. *Biological Conservation*, 72, 387–391.

(3) Wang J., Wu X.B., Tian D., Zhu J., Wang R. & Wang C. (2011) Nest-site Use by the Chinese alligator (*Alligator sinensis*) in the Gaojingmiao breeding farm, Anhui, China. *Asian Herpetological Research*, 2, 36–40.

(4) Sanchez M. (2012) Mitigating habitat loss by artificial egg laying sites for Reunion day gecko *Phelsuma borbonica*, Sainte Rose, Reunion Island. *Conservation Evidence*, 9, 17–22.

(5) Paterson J.E., Steinberg B.D. & Litzgus J.D. (2013) Not just any old pile of dirt: evaluating the use of artificial nesting mounds as conservation tools for freshwater turtles. *Oryx*, 47, 607–615.

(6) Wnek J.P., Bien W.F. & Avery H.W. (2013) Artificial nesting habitats as a conservation strategy for turtle populations experiencing global change. *Integrative Zoology*, 8, 209–221.

(7) Quinn D.P., Kaylor S.M., Norton T.M. & Buhlmann K.A. (2015) Nesting mounds with protective boxes and an electric wire as tools to mitigate diamond-backed terrapin (*Malaclemys terrapin*) nest predation. *Herpetological Conservation & Biology*, 10, 969–977.

(8) Nagle R.D. & Congdon J.D. (2016) Reproductive ecology of *Graptemys geographica* of the Juniata river in Central Pennsylvania, with recommendations for conservation. *Herpetological Conservation and Biology*, 11, 232–243.

(9) Wilson B., Grant T.D., Van Veen R., Hudson R., Fleuchaus D., Robinson O. & Stephenson K. (2016) The Jamaican Iguana (*Cyclura collei*): A report on 25 years of conservation effort. *Herpetological Conservation and Biology*, 11, 237–254.

13.13. Create or restore ponds

- **Four studies** evaluated the effects of creating or restoring ponds on reptile populations. Two studies were in the USA[2,3] and one was in each of Austria[1] and China[4].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (4 STUDIES)

- **Use (4 studies):** Four studies (including one replicated and one before-and-after study) in Austria[1], the USA[2,3] and China[4] reported that following the creation of ponds[2,4], in one case 30–60 years after pond creation[3], or restoration of a river island that included creation of ponds[1] grass snakes and sand lizards were found on the island[1], and ponds were occupied by mangrove salt marsh snakes[2], common snapping turtles and midland painted turtles[3] and Chinese alligators[4].

Background

Ponds are often drained, left to dry or degraded during the development of agriculture or expansion of urban areas or other land uses. Ponds created to improve wildlife habitat, e.g. for fish or frogs, may also used by reptiles such as snakes and turtles (Adams & Saenz 2011). Restoration of ponds may therefore help to increase populations of reptiles that are dependent on ponds. Ponds may be restored (or maintained to prevent desiccation) by deepening, de-silting, or re-profiling activities, planting new or managing existing vegetation to provide appropriate shading levels and basking sites for reptiles, and adding woody debris.

For studies discussing wetlands and waterway management, see *Create or restore wetlands* or *Create or restore waterways*.

Adams C.K. & Saenz D. (2011) Use of artificial wildlife ponds by reptiles in eastern Texas. *Herpetological Bulletin,* 115, 4–11.

A before-and-after study in 1998 of constructed ponds and restructured shoreline of the constructed Danube Island, Austria (1) found that in the first year, two of nine species found in the surrounding area had colonised the island. Two of nine reptile species (grass snake *Natrix natrix* and sand lizard *Lacerta agilis*) recorded in the broader locality were observed on the island at some of the newly created inshore zones. The 21 km shoreline, which was straight with steep embankments, was restructured by creating shallow water areas, gravel banks, small permanent backwaters and temporary waters. Thirteen newly-created inshore zones and existing artificial water bodies (created 1989–1997) and one natural water body were monitored for reptile colonization. Monitoring was undertaken during 20–32 visits (day and night) in February–October 1998 by visual surveys.

A study in 2006–2007 in two artificial ponds in Florida, USA (2) found that both ponds were occupied by mangrove salt marsh snakes *Nerodia clarkia compressicauda*, though the number of snakes may have decreased following herbicide application to ponds. In spring 2006, it was estimated that there were 33 mangrove salt marsh snakes in the artificial ponds and in spring 2007 there were 95 snakes. In summer 2007, while glyphosate herbicide was being applied to the pond vegetation, there were 94 snakes, but numbers were estimated at 47 snakes that autumn. Two artificial ponds (0.17–0.43 ha) were created to collect stormwater run-off in 1996 and 2004. Both ponds were treated with glyphosate herbicide ('Aquamaster') monthly during summer 2007 (exact start date not known). Salt marsh snakes were monitored at night for three nights at a time in April 2006 (spring), March–April 2007 (spring), May–July 2007 (summer) and August–October 2007 (autumn). Snakes were caught by hand, individually marked with PIT tags and released. Snakes >40 cm long were included in the calculation of abundance.

A replicated study in 1957–1980 and 2005–2013 in mixed oak forest and agricultural land in Pennsylvania, USA (3) found that 30–60 years after eight artificial ponds were created, two aquatic turtle species were present in all almost all ponds. Approximately 30–60 years after eight artificial ponds were created, common snapping turtles *Chelydra serpentina serpentina* were present at all eight ponds and midland painted turtles *Chrysemys picta marginata* were present at six of eight ponds. In one pond, both adult and juvenile individuals were caught of both species. The authors reported that common snapping turtles first colonised one pond three years after it was created and midland painted turtles were first recorded in the same pond 16 years after it was created. In 1957–1980, eight artificial ponds (872–5,989 m^2) were constructed on either side of the boundary of a nature reserve (856 ha). Turtles were monitored using two baited hoop-nets/pond for two consecutive sets of five-day trapping periods in June–July 2005–2013. One pond was monitored for turtles in 2005–2013 and the remaining seven ponds were monitored for turtles in 2013 only. All turtles were individually marked prior to release.

A replicated study in 2006–2016 in an area of ponds and dense vegetation in Anhui Province, China (4) found that around half of constructed ponds were used by Chinese alligators *Alligator sinensis* following the release of captive bred individuals. Alligators were distributed among 28 of 50 constructed ponds. Successful reproduction was recorded two years after the first release (158 eggs, producing 80 hatchlings were discovered), though the full extent of nesting was unknown. Fifty ponds (30 ha total water area) were constructed in the release area, at a cost of around $US10,000/pond. Ponds were established with terrestrial (e.g. bamboo) and aquatic vegetation, and "seeded" with fish, amphibians and snails. In 2006–2016, eleven releases (during May–June) of 93 alligators were carried out (sex ratio 1 male:2 females) and population monitoring was carried out using spotlight surveys.

(1) Chovanec A., Schiemer F., Cabela A., Gressler S., Grotzer C., Pascher K., Raab R., Teufl H. & Wimmer R. (2000) Constructed inshore zones as river corridors through urban areas - the Danube in Vienna: preliminary results. *Regulated Rivers-Research & Management*, 16, 175–187.

(2) Ackley J.W. & Meylan P.A. (2010) Watersnake eden: Use of stormwater retention ponds by mangrove salt marsh snakes (*Nerodia clarkii compressicauda*) in urban Florida. *Herpetological Conservation and Biology*, 5, 17–22.

(3) Hughes D.F., Tegeler A.K. & Meshaka Jr W.E. (2016) Differential use of ponds and movements by two species of aquatic turtles (*Chrysemys picta marginata* and *Chelydra serpentina serpentina*) and their role in colonization. *Herpetological Conservation and Biology*, 11, 214–231.

(4) Manolis C., Shirley M., Siroski P., Martelli P., Tellez M., Meurer A. & Merchant M. (2016) *CSG Visit to China, August 2016*. IUCN-SSC Crocodile Specialist Group.

13.14. Create or restore rock outcrops

- **Five studies** evaluated the effects of creating or restoring rock outcrops on reptile populations. All five studies were in Australia[1-5].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Abundance (1 study):** One replicated, paired sites, controlled study in Australia[3] found that areas restored with artificial rocks had a higher abundance of adult velvet geckos and similar numbers of juveniles compared to unrestored areas.
- **Survival (1 study):** One replicated, paired sites, controlled study in Australia[3] found that in areas restored with artificial rocks, juvenile velvet geckos had higher survival rates than in unrestored areas.

BEHAVIOUR (4 STUDIES)

- **Use (4 studies):** One replicated, randomized, controlled study in Australia[4] found that some restored rocky outcrops were recolonized by pink-tailed worm-lizards. One replicated, controlled study in Australia[5] found that constructed rock outcrops were used by two snake and six lizard species at least as often as natural outcrops. Two replicated studies (including one randomized study) in Australia[1,2] found that artificial rock outcrops were used by two lizard and one snake species[1] and six lizard and two snake species[2]. One study[1] also found that unshaded artificial rocks were used more frequently by velvet geckos than shaded ones.

Background

Rock outcrops are home to a diverse range of highly specialised species (Goldingay & Newall 2017) but may be removed or degraded. Retaining rocky outcrops or structures is important for rock-dwelling reptiles as rocks provide a variety of microhabitats, including crevices, overhangs and suitable temperature shelter sites (Michael *et al.* 2010). Where rocks or rock piles have been removed they may be rebuilt and restored, and sometimes artificial rocks are used.

For studies that provide artificial cover or artificial shading for reptiles, see *Create artificial refuges, hibernacula and aestivation sites* or *Threat: Climate change and severe weather – Provide artificial shade for individuals* and *Provide artificial shade for nests or nesting sites.*

Michael D.R., Lindenmayer D.B. & Cunningham R.B. (2010) Managing rock outcrops to improve biodiversity conservation in Australian agricultural landscapes. *Ecological Management & Restoration*, 11, 43–50.

Goldingay R.L. & Newell D.A. (2017) Small-scale field experiments provide important insights to restore the rock habitat of Australia's most endangered snake. *Restoration Ecology*, 25, 243–252.

A replicated, randomized study in 1994–1995 on a sand plateau in New South Wales, Australia (1) found that reptiles used artificial rocks (concrete pavers/paving stones) and tended to be found more often under unshaded artificial rocks with narrow crevices. Velvet geckos *Oedura lesueurii* used 28 unshaded pavers (45 individuals recorded) and nine shaded pavers (11 individuals recorded), of which 26 pavers were narrow-creviced (44 individuals recorded) and 12 were wide-creviced (12 individuals recorded). One skink *Cryptoblepharus virgatus* and one broad-headed snake *Hoplocephalus bungaroides* were recorded in one unshaded, narrow-creviced paver each. In November 1994–January 1995, artificial rocks (square concrete pavers: 19 cm wide, 5 cm thick) were placed in groups of four (20 cm apart in a square formation) at three undisturbed rock outcrops (sites >1 km apart, 32–52 total pavers/site). Rocks were modified with either 4 mm or 8 mm crevices (created by gluing wood to the underside of the pavers) and unshaded or shaded (90 x 50 cm steel frame covered with two layers of shade cloth; unshaded pavers had only steel frames). Surveys were attempted six times/site in April–November 1995 (18 total surveys) with reptiles marked with a toe clip. Human disturbance of artificial rocks prevented seven of 18 surveys from being carried out.

A replicated study in 2007–2008 on two sandstone plateaus in New South Wales, Australia (2) found that most artificial rocks were colonized by reptiles within 40 weeks. Artificial rocks started to be colonised by reptiles after six weeks. After 14 weeks, 50% of rocks were used and after 40 weeks, 82% were used. Lizards began using rocks after six weeks and snakes by 28 weeks (six lizard and two snake species were recorded in total). Rock spacing (either placed > 3 m from other rocks, or in pairs separated by < 0.5 m) did not affect colonisation rates (data reported as model outputs). The daily thermal characteristics (maximum, minimum and range of temperatures) of artificial rocks were similar to natural rock (see paper for details). In July–August 2007 artificial rocks (198 fibre-reinforced cement 55 x 39 x 4 cm with crevices constructed on the bottom) were placed at five sites (20 at each of two sites with no natural rock removal in a national park, and 40–72 at three sites with rock removal). Reptiles were surveyed on artificial and natural rocks 14 times in July 2007–May 2008 by turning rocks by hand.

A replicated, paired sites, controlled study in 2009–2010 on a sand plateau in New South Wales, Australia (3) found that sites restored with artificial rocks had higher abundances of adult, but not juvenile, velvet geckos *Oedura lesueurii* compared to unrestored sites, but juveniles had higher survival rates in restored sites. Adult gecko abundances were higher in sites restored by adding artificial rocks (12–23 individuals/site) compared to unrestored sites (2–7), whereas juvenile abundances tended to be similar (restored: 37, unrestored: 18). Juvenile survival rates were higher in restored (90% survival) than unrestored sites (80%), whereas adult survival rates were similar (restored: 92–93%, unrestored: 78–92%). Gecko abundances were similar underneath individual artificial and individual natural rocks (results reported as model outputs). Six rock outcrop sites were restored by adding 50 artificial rocks (fibre-reinforced cement 51 x 35 x 5 cm) to natural rocks (18 natural rocks on average/site). Each restored site was paired with a nearby (average 500 m apart) unrestored site (26 natural rocks). Reptiles were surveyed by turning all artificial and natural rocks and hand-capturing geckos monthly throughout 2009–2010. Geckos were toe clipped before being released.

A replicated, randomized, controlled study in 2014–2015 in six rock and grassland areas in Australian Capital Territory, Australia (4) found that when rock cover and native grasses were restored, Australian pink-tailed worm-lizards *Aprasia parapulchella* recolonised rock outcrops within one year, depending on additional management. Results were not statistically tested. Pink-tailed worm-lizards recolonised restored rock cover and grassed plots after nine months (rock and plants restored only: 4 live lizards and 1 shed skin observed; rock and plants restored plus prescribed fire and herbicide application: 4 live lizards; rock and plants restored plus prescribed fire: 2 lizards). There was no evidence of lizards at unrestored sites of poor habitat quality or at sites with rock and plant restored combined with herbicide application only. Four lizards and three shed skins were observed in plots in unrestored, nearby high-quality lizard habitat. In April–May

2014, plots (4 x 4 m) in six sites (150 m apart) were managed by: rock (30% rock cover) and native grass restoration alone; rock and grass restoration with prescribed fire (using a blow torch); rock and grass restoration with herbicide application (Glyphosate, 1:100 glyphosate:water); or rock and grass restoration with prescribed fire and herbicide application. In each site, two additional plots received no rock or plant restoration (one was adjacent to managed plots and the second was in nearby high-quality lizard habitat). In February 2015, all plots were surveyed for lizards (live sightings and skins) including two unmanaged plots/site (one in poor, the other near high-quality lizard habitat).

A replicated, controlled study in 1998–2013 of rocky outcrops in the southern metropolitan area of Sydney, Australia (5) found that constructed outcrops were occupied by broad-headed snakes *Hoplocephalus bungaroides*, small-eyed snakes *Cryptophis nigrescens*, velvet geckos *Oedura lesueurii* and five species of skink at least as often as natural outcrops. Broad-headed snakes and small-eyed snakes were recorded in a similar proportion of constructed outcrops (broad-headed: 49%; small-eyed: 27%) as natural outcrops (broad-headed: 48%; small-eyed: 52%). Velvet geckos and skinks (five species grouped together) were more abundant in constructed outcrops (1.6 geckos/10 rocks, 0.6 skinks/10 rocks) than natural outcrops (0.7 geckos/10 rocks, 0.2 skinks/10 rocks). The authors reported that broad-headed snakes were more likely to be recorded in outcrops >500 m from trails or roads (75% probability) than <150 m from trails or roads (41% probability, see original paper for details). In March 1998 and 1999, thirty-three outcrops were constructed in an area of a national park (8 x 10 km) by placing rocks on a rocky platform in a grid (22 small 10-rock/platform outcrops and six pairs of large (12 total) 50-rock/platform outcrops, see original paper for details). In total 33 constructed outcrops (one small outcrop was excluded from analysis) and 31 natural outcrops were surveyed for reptiles during August–September in seven years between 2000–2013 (starting 1–2 years after outcrops were constructed). Reptiles were monitored by lifting rocks to reveal any inhabitants.

(1) Webb J.K. & Shine R. (2000) Paving the way for habitat restoration: can artificial rocks restore degraded habitats of endangered reptiles? *Biological Conservation*, 92, 93–99.

(2) Croak B.M., Pike D.A., Webb J.K. & Shine R. (2010) Using artificial rocks to restore nonrenewable shelter sites in human-degraded systems: colonization by fauna. *Restoration Ecology*, 18, 428–438.

(3) Croak B.M., Webb J.K. & Shine R. (2013) The benefits of habitat restoration for rock-dwelling velvet geckos *Oedura lesueurii*. *Journal of Applied Ecology*, 50, 432–439.

(4) McDougall A., Milner R.N.C., Driscoll D.A. & Smith A.L. (2016) Restoration rocks: integrating abiotic and biotic habitat restoration to conserve threatened species and reduce fire fuel load. *Biodiversity and Conservation*, 25, 1529–1542.

(5) Goldingay R.L. & Newell D.A. (2017) Small-scale field experiments provide important insights to restore the rock habitat of Australia's most endangered snake. *Restoration Ecology*, 25, 243–252.

Whole habitat restoration

13.15. Restore island ecosystems

- **Three studies** evaluated the effects of restoring island ecosystems on reptile populations. One study was in each of the Seychelles[1], the USA[2] and the US Virgin Islands[3].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (3 STUDIES)

- **Occupancy/range (1 study):** One study in the US Virgin Islands[3] found that following translocation to a restored island, St. Croix ground lizards expanded their range during the fifth to seventh year after release.
- **Reproductive success (2 studies):** One study in the Seychelles[1] found that following a range of interventions carried out to restore an island ecosystem, the number of hawksbill and green turtle nests increased. One replicated study in the USA[2] found that during and after an island was rebuilt, diamondback terrapins continued to nest on the island.

BEHAVIOUR (0 STUDIES)

Background

Islands account for approximately 5% of the earth's landmass. They are home to high levels of biodiversity; however biodiversity is disproportionately threatened on islands compared to continental areas, with 61% of all extinct and 37% of all critically endangered species endemic to islands (Tershy *et al.*, 2015). Restoration efforts often incorporate multiple actions such as: native vegetation planting, removal of invasive species, beach restoration and translocating native or surrogate fauna to re-establish ecosystem function across the whole island. The outcomes of island restoration are often realised over long timescales and study results should be considered with this in mind.

This action includes studies that have carried out a large combination of different interventions to restore the whole island ecosystem, including both plant and animal communities.

Studies discussing the effectiveness of individual actions or actions carried out on only parts of an island are covered elsewhere. For example, replanting vegetation (*Plant native species*), removing invasive or problematic species (*Threat: Invasive or problematic species*), or species reintroductions (*Species Management*).

Tershy B.R., Shen K.W., Newton K.M., Holmes N.D. & Croll D.A. (2015) The importance of islands for the protection of biological and linguistic diversity. *Bioscience*, 65, 592–597.

A study in 1992–2006 on a tropical island in Seychelles (1) found that a programme of island restoration, including a large range of measures such as eradicating many invasive, non-native species and measures to control poaching, resulted in an increase in the number of hawksbill *Eretmochelys imbricata* and green turtle *Chelonia mydas* nests. Between 22 and 25 years after the start of an island restoration programme there were 19–43 sea turtle nests/year and 25–35 years after the programme's anti-poaching measures were introduced, there were 66–108 sea turtle nests/year. The authors reported that the number of sea turtle nests had increased in each year of the study. In 1970s–2000s, Cousine Island (27 ha) underwent restoration, including invasive plant and animal removal, introduction of poaching controls and anti-poaching initiatives (details not provided), reintroducing native plants and bird species, increased biosecurity measures for incoming goods and the confinement of agricultural plants to a designated area (see original paper for details on all measures undertaken). Sea turtle nests were monitored from the 1990s onwards (no details were provided).

A replicated study in 2002–2011 in beaches on Poplar Island, Maryland, USA (2) found that during the island rebuilding process, diamondback terrapins *Malaclemys terrapin* continued to nest on the island. Two years after island rebuilding began, 68 nests were laid on the island compared to 211 nests laid 11 years after rebuilding began. The highest number observed (282 nests) were laid five years after rebuilding began. Nest survival rates ranged from 59–85% over the period of 3–11 years after rebuilding began. Poplar Island was rebuilt from three 4 ha remnants starting in 2000 using the footprint of the island from 1850 (450 ha). A perimeter dyke was constructed in 2002 and the interior began to be filled with stone and dredged sand (expected completion in 2027). Nesting areas were monitored daily, and nests marked with flagging and covered with hardware cloth (1.25 cm^2 mesh) to prevent bird predation. After 45–50 days, the hardware cloth was removed, and a metal flashing ring buried 10 cm around nests to capture hatchlings.

A study in 2013–2015 on a mixed forest and scrubland island in the US Virgin Islands (3) found that St. Croix ground lizards *Ameiva polops* translocated to a restored island continued to increase their range annually in the fifth to seventh year after being released. Five years after St. Croix ground lizards were released, lizards occupied 41% of sites surveyed and 69% of sightings were <200 m from the release site. Six years after release, lizards occupied 60–66% of sites surveyed and seven years after release this increased to 74–87% of sites surveyed. Lizards recolonised the island from west to east (see original paper for details). Restoration of native habitat, including forest, woodland, scrubland and sandy beaches, had been underway on Buck Island (71 ha) for 40 years prior to lizards being released in 2008. A total of 57 lizards were introduced to the island in 2007 and population surveys were carried out in 63 sites (1,260 m^2 circular sites, at least 80 m apart). Sites were surveyed for three days, five times/season in May 2013, May 2014, October 2015, May 2015 and October 2015. In addition, in May 2013 a total of 192 extra surveys were carried out in 32 sites, which were surveyed twice a day for three consecutive days.

(1) Samways M.J., Hitchins P.M., Bourquin O., Henwood J. (2010) Restoration of a tropical island: Cousine Island, Seychelles. *Biodiversity and Conservation*, 19, 425–434.

(2) Roosenburg W.M., Spontak D.M., Sullivan S.P., Matthews E.L., Heckman M.L. Trimbath R.J., Dunn R.P., Dustman E.A., Smith L. & Graham L.J. (2014) Nesting habitat creation enhances recruitment in a predator-free environment: *Malaclemys* nesting at the Paul S. Sarbanes Ecosystem Restoration Project. *Restoration Ecology*, 22, 815–823.

(3) Angeli N.F., Lundgren I.F., Pollock C.G., Hillis-Starr Z.M. & Fitzgerald L.A. (2018) Dispersal and population state of an endangered island lizard following a conservation translocation. *Ecological Applications*, 28, 336–347.

13.16. Create or restore grasslands

- **Four studies** evaluated the effects of creating or restoring grasslands on reptile populations. One study was in each of South Africa[1] China[2], Australia[3] and the USA[4].

COMMUNITY RESPONSE (2 STUDIES)

- **Richness/diversity (2 studies):** One replicated, controlled study in Mongolia[2] found that areas of restored grassland had similar species richness compared to unrestored areas. One replicated, site comparison study in South Africa[1] found that an area of restored grassland had lower species richness than natural grassland in three of four comparisons.

POPULATION RESPONSE (3 STUDIES)

- **Abundance (3 studies):** One of two replicated, controlled studies (including one paired study) in Mongolia[2] and the USA[4] found that areas of restored grassland had higher lizard abundance than unrestored areas[2]. The other study[4] found that areas of restored grassland had fewer snakes than unrestored areas. One replicated, site comparison study in South Africa[1] found that an area of restored grassland had a similar abundance of reptiles compared to two areas of natural grassland.

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One replicated, randomized, controlled study in Australia[3] found that some areas of restored grassland and rocky outcrops were recolonized by pink-tailed worm-lizards.

Background

Many grasslands have been lost to agricultural intensification through conversion to cropland or through agricultural abandonment, which may result in colonization by scrub or woodland. Agri-environment schemes in Europe and North America support the preservation or restoration of grasslands for agricultural, conservation and carbon storage reasons. Restoration of these grasslands may benefit some reptile species that are associated with them.

See also: *Create or* restore *savannas*.

A replicated, site comparison study in 2005–2006 in an area of high altitude grassland in Gauteng, South Africa (1) found that in an area of restored grassland that was previously cultivated, reptile species richness was lower in three of four comparisons than in areas of natural grassland, but total number of reptiles caught was similar. Reptile species richness was lower in previously cultivated grassland (7 species) than in natural grassland with no rocks (13 species) or many rocks (12 species) in three of four comparisons (other comparison found no significant difference; see paper for details). Total reptile captures was similar in all three grassland habitats (previously cultivated: 31 captures; natural no rocks: 66 captures; natural with rocks: 53 captures). In 2005, a nature reserve was expanded to include three areas of previously cultivated land (18,600 ha in total).

One area was last ploughed in 2000–2002; the second was last cultivated in 2002–2003; the third area was ploughed in 2005 to allow for reseeding with indigenous species (details of reseeding not provided). These areas were compared to two areas of natural grassland, one of which had an abundance of scattered rocks. In December 2005–April 2006, a total of nine groups of traps (36 m drift fence, 5 pitfall and 8 funnel traps) were set up in the three habitat types (3 groups/habitat type). Traps were checked daily and all reptiles were identified to species level and rereleased where they were caught.

A replicated, controlled study in 2012 in sandy steppe grassland in Inner Mongolia, China (2) found that restored grassland with dense vegetation had similar lizard species richness but greater abundance than degraded sparse grassland or natural grassland. Lizard species richness was similar but abundance was greater in restored grassland (richness: 3 species/plot, abundance: 58 individuals/plot) compared to degraded sparse, cash-crop dominated grassland (richness: 3, abundance: 42) and natural grassland (richness: 3, abundance: 28). Lizards were surveyed in 10 plots in each of three types of grassland: grassland restored to combat desertification and dominated by sweet vetch *Hedysarum* spp., korshinsk peashrub *Caragana korshinskii* and erect milkvetch *Astrogalus adsurgens*; degraded sparse grassland dominated by cash crops such as herba ephedra *Ephedra sinica* and alfalfa *Medicago sativa*, or natural (undisturbed) grassland. Lizards were surveyed using drift fences with pitfall traps (eight traps/plot) over seven consecutive days in June, July and September 2012 (21 trap days/plot). Lizards were individually marked by toe clipping prior to release.

A replicated, randomized, controlled study in 2014–2015 in six rock and grassland areas in Australian Capital Territory, Australia (3) found that when native grasses and rock cover were restored, Australian pink-tailed worm-lizards *Aprasia parapulchella* recolonised rock outcrops within one year depending on additional management. Results were not statistically tested. Pink-tailed worm-lizards recolonised restored grass and rock plots after nine months (grass and rocks restored only: 4 live lizards and 1 shed skin observed; grass and rocks restored plus prescribed fire and herbicide application: 4 live lizards; grass and rocks restored plus prescribed fire: 2 lizards). There was no evidence of lizards at unrestored sites of poor habitat quality or at sites with grass and rock restored combined with herbicide application only. Four lizards and three shed skins were observed in plots in unrestored, nearby high-quality lizard habitat. In April–May 2014, plots (4 x 4 m) in six sites (150 m apart) were managed by: native grass and rock (30% rock cover) restoration alone; grass and rock restoration with prescribed fire (using a blow torch); grass and rock restoration with herbicide application (Glyphosate, 1:100 glyphosate:water); or grass and rock restoration with prescribed fire and herbicide application. In February 2015, all plots were surveyed for lizards (live sightings and skins) including two unmanaged plots/site (one in poor, the other near high-quality lizard habitat).

A replicated, paired sites, controlled study in 2014–2015 in four grasslands in California, USA (4) found that restored grasslands had reduced snake abundance 13–24 years after restoration took place. After 13–24 years, restored grassland had 10 times lower snake abundance (0.09 snakes/plot) than unrestored grassland (0.92 snakes/plot). The authors reported that snake abundance was correlated with abundance of non-native house mice *Mus musculus.* In 1992 and

2003, four grasslands were partially restored with native perennial plants. In 2014–2015, two paired survey blocks (each 150 x 150 m) were set up in each grassland (8 total blocks): restored grassland and exotic annual grassland. Snake monitoring was carried out in April, July, and November 2014 and February–March 2015. Snakes were surveyed using three pairs of coverboards (metal and plywood) spaced at 75 m intervals along 150 m transects (4 transects/block, 192 total coverboards). Coverboards were surveyed in the mornings up to eight times/block and season (3,216 total coverboard surveys).

(1) Masterson G.P., Maritz B., Mackay D. & Alexander G.J. (2009) The impacts of past cultivation on the reptiles in a South African grassland. *African Journal of Herpetology*, 58, 71–84.

(2) Zeng Z.-G., Bi J.-H., Li S.-R., Chen S.-Y., Pike D.A., Gao Y. & Du W.-G. (2014) Effects of habitat alteration on lizard community and food web structure in a desert steppe ecosystem. *Biological Conservation*, 179, 86–92.

(3) McDougall A., Milner R.N.C., Driscoll D.A. & Smith A.L. (2016) Restoration rocks: integrating abiotic and biotic habitat restoration to conserve threatened species and reduce fire fuel load. *Biodiversity and Conservation*, 25, 1529–1542.

(4) Wolf K.M., Whalen M.A., Bourbour R.P. & Baldwin R.A. (2018) Rodent, snake and raptor use of restored native perennial grasslands is lower than use of unrestored exotic annual grasslands. *Journal of Applied Ecology*, 55, 1133–1144.

13.17. Create or restore savannas

- **One study** evaluated the effects of creating or restoring savannas on reptile populations. This study was in Australia[1].

COMMUNITY RESPONSE (1 STUDY)

- **Richness/diversity (1 study):** One before-and-after study in Australia[1] found that reptile species richness was higher following restoration of savanna-like habitat on a golf course.

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (0 STUDIES)

Background

Through under-grazing or burning suppression, savanna vegetation can revert to denser scrubland. Restoring savanna may benefit reptiles typically associated with the habitat.

See also: *Create or restore grassland*.

A before-and-after study in 1996–2004 of a golf course with degraded savanna-like habitat of open woodland and grassland in Sydney, Australia (1) found that restoration that included leaving unmown buffers around ponds, removing non-native weeds, planting native vegetation and adding woody debris resulted in an increase in reptile species over eight years. Reptiles increased from three to eight species in the first two years and to nine species after five years, then remained stable for the following three years. A total of 37 reptile species were predicted in the area of which eight were present following restoration (in 2004), compared to three prior to restoration (in 1996). The golf course was developed in 1993 and restoration undertaken in 1997–2001. The mowing regime was changed to develop rough grassland and a narrow band of herb vegetation was retained around ponds as a buffer zone; native shrubs and trees were planted; non-native weeds were removed and coarse woody debris was reintroduced onto the woodland floor. Reptile surveys were carried out by visual searches (4 hrs long) and checking 12 artificial shelters once a season in 1996–2004.

(1) Burgin S. & Wotherspoon D. (2009) The potential for golf courses to support restoration of biodiversity for biobanking offsets. *Urban Ecosystems*, 12, 145–155.

13.18. Create or restore forests

- **Six studies** evaluated the effects of creating or restoring forests on reptile populations. Three studies were in the USA[2,3,6], two were in Australia[1,5] and one was in Mexico[4].

COMMUNITY RESPONSE (3 STUDIES)

- **Richness/diversity (3 studies):** One of two replicated studies (including one randomized, controlled study) in the USA[2] and Australia[5] found that restored and natural riparian forest had similar reptile species richness[2]. The other study[5] found that restored forest areas had higher reptile species richness than remnant forest areas. One replicated, site comparison study in Australia[1] found that the type of restoration had mixed effects on reptile species richness in tropical and subtropical areas.

POPULATION RESPONSE (5 STUDIES)

- **Abundance (5 studies):** Two of three replicated studies (including two controlled, before-and-after studies) in the USA[3,6] and Mexico[4] found that areas of restored forest had similar abundances of snakes[4] and six lizard species[3] as unrestored areas. The other study[6] found that restoring forest stands had mixed effects on the abundance of reptiles. One replicated, site comparison study in Australia[1] found that areas with different restoration types had similar reptile abundance in tropical and subtropical areas. One replicated, randomized, controlled study in Australia[5] found that restored forest areas had higher reptile abundance than remnant forest areas.

BEHAVIOUR (0 STUDIES)

Background

Restoring or creating forest and woodland may provide important habitat for forest-dependant reptile species, particularly in disturbed or fragmented landscapes. Trees grow slowly and therefore the effects of forest restoration may not be evident for decades or even longer after restoration begins. Care must therefore be taken when interpreting the results of these studies.

For studies on other actions relating to forest management, see *Threat: Biological resource use – Logging and wood harvesting*.

A replicated, site comparison study in 2000–2001 in tropical and subtropical rainforests in Queensland and New South Wales, Australia (1) found that overall reptile richness, but not abundance, varied by restored forest type, depending on the region and species' habitat specialism. In the tropics, management type affected overall reptile species richness (ecological restoration: 0.9–1.0 species/site, mixed timber plantation: 0–0.4, young monoculture plantation: 0–

1.4, old monoculture plantation: 0.1–1.5, natural regrowth: 0–0.4, converted pasture: 0–0.01, old-growth forest: 0.1–2.2) but not abundance (restoration: 9.6 individuals/site, mixed: 2.8, young: 10.4, old: 6.0, regrowth: 0.8, pasture: 0.5, old-growth: 8.8). In the subtropics, management type did not affect overall species richness (restoration: 0–1.0, mixed: 0–0.7, young: 0–0.6, old: 0.2–0.4, regrowth: 0.2–0.4, pasture: 0–0.2, old-growth: 0.4–1.3) or abundance (restoration: 13.6, mixed: 10.7, young: 2.4, old: 1.3, regrowth: 17.6, pasture: 0.3, old-growth: 4.0). Rainforest-specialist species richness varied by management type in both tropical and subtropical regions and were only recorded in restoration plantings, old plantations, and old-growth forest in the tropics and in young and old plantations, natural regrowth and old-growth forest in the subtropics (see paper for individual species results). Reptiles were monitored in ecological restoration plantings (19 sites), mixed timber plantations (15), young monoculture timber plantations (10), old monoculture timber plantations (20), natural regrowth (10), converted to pasture (10), and unmanaged old growth rainforest (20) in subtropical and tropical rainforest. Visual searches were carried out in one 0.3 ha plot/site (30 minutes/search) on three occasions/site between October 2000–November 2001.

A replicated, site comparison study in 1999–2000 of five riparian forest sites in California, USA (2) found that reptile species richness in restored riparian forest was similar to that in natural riparian forest. Similar numbers of reptile species (4 species) were found in restored riparian forest compared to natural riparian forest (data reported as statistical model outputs). The authors reported that species abundant in the restored sites tended to be generalist species (e.g. coast garter snakes *Thamnophis elegans terrestris*) and that forest specialists (e.g. northern alligator lizards *Elgaria coerulea*) were present in the natural forest but not in the restored forest. Restoration, which included planting of woody riparian species, commenced between 1996 and 1998. In 1996–1998, a total of 15 ha of woody riparian species and 2.4 ha of freshwater wetland species were planted. Three restored sites (17,400 m^2, 28,000 m^2, 65,000 m^2) were compared to two mature riparian forest sites (47,420 m^2 and 24,780 m^2). Reptiles were sampled using pitfall traps during May–August 2000 and visual surveys (25 x 25 m area).

A replicated, controlled, before-and-after study in 2000–2006 in three sites of riparian forest in central New Mexico, USA (3, likely same experimental set-up as 4) found that restoring forest through removing non-native vegetation and either burning the removed vegetation or planting native shrubs resulted in no change in the abundance of six lizard species. The effect of burning the removed vegetation and planting native shrubs cannot be separated from the effect of vegetation removal. Over a period of 1–3 years since removal, abundance of the six most common lizards (5 other species detected but not included in analysis due to small sample sizes) remained similar for restored and unmanaged sites (data reported as statistical model outputs). In 2003–2005, four riparian sites each within three regions were selected for non-native vegetation removal (3 sites/region) or no vegetation removal (1 site/treatment). Removal consisted of mechanical removal with chainsaws and herbicide (Garlon) application to stump sprouts. One removal site/region also had all removed vegetation burned, and another also had native shrubs planted. In June–September 2001–2006, abundance of lizards was surveyed at all sites with drift-fencing, pitfall and funnel traps (3 trapping arrays/site, checked 3 times/week).

A replicated, controlled, before-and-after study in 2000–2006 in three areas of mixed riparian forest in north, middle and south Mexico (4, likely same experimental set-up as 3) found that restoring forest through removing non-native vegetation and either planting native shrubs or burning slash piles did not increase overall snake abundance. The effect of planting native shrubs and burning slash piles cannot be separated from the effect of vegetation removal. Snake abundance remained similar in restored and unmanaged sites (data reported as statistical model outputs). Fourteen species of snake were counted in the sites over seven years of surveys. Snakes were monitored in 12 sites (20 ha each) in 2000–2006 from three areas of forest (four sites/area). In 2003–2005, the sites in each area were managed by either removing non-native plants (using chainsaws and herbicide), or removing non-native plants and planting native shrubs, or removing non-native plants and burning slash piles, or not managed at all (see original paper for details). Snakes were monitored using drift fences with pitfall and tunnel traps ('arrays'; 3 arrays/site) in June–July 2000 and June–September 2001–2006.

A replicated, randomized, controlled study in 2011–2012 in upland forest in Queensland, Australia (5) found that reptile captures and species richness tended to be higher in restoration plantings than remnant forest, particularly when coarse woody debris was added. Results were not statistically tested. Reptile captures and species richness tended to be highest in restoration plantings with added coarse woody debris (captures: 3.7–4.0 individuals/site; species richness: 2.0 reptiles/site), followed by restoration plantings without added coarse woody debris (1.5, 0.7), and lowest in remnant forest without added debris (0.8, 0.5) or remnant forest with coarse woody debris removed (0.3, 0.2). In November 2011–January 2012, five treatments were applied four times each in four sites (60 m x 40 m sites): restoration planting (native trees and shrubs) with added salvaged log piles; restoration planting with added fence post piles; restoration planting with no debris added; remnant forest with no debris added; and remnant forest with all woody debris removed. Restoration plantings were 0–7 years old when coarse woody debris was added. Reptiles were surveyed in either March or August 2012 and again in December 2012.

A replicated, site comparison study in 2012–2014 in saltcedar *Tamarix ramosissima*-cottonwood *Populus fremontii* forest along a river in Utah, Arizona and Nevada, USA (6) found that restoring forest stands through replanting native species, managing vegetation using cutting and herbicides, and redirecting water flow to reduce dominance of invasive saltcedar had mixed effects on overall lizard abundance. Trapping surveys indicated that overall lizard abundance was similar in restored stands (127–171 lizards/site/100 trap nights) compared to unrestored stands (62–74), whereas visual encounter surveys found that overall reptile abundance was greater at restored sites (results reported as statistical tests). See original paper for the effects of restoration on individual species. In winter–spring 2012–2013, restoration of saltcedar-cottonwood/willow *Salix* spp stands was carried out along the Virgin River, including: mechanically removing 50% of saltcedar and Russian olive *Elaegnus angustifolia*, spraying stumps with herbicide, transplanting native plants and introducing/redirecting water flows by trenching. Saltcedar in Utah was subject to biocontrol by northern tamarisk beetles *Diorhabda carinulata* from 2006 (see original paper for details). Reptiles were monitored in two restored and six unrestored stands in May–July 2013–2014 using drift fences with pitfall and funnel traps (1,060 total trap days) and visual encounter surveys (3 transects/site, see original paper for details).

(1) Kanowski J.J., Reis T.M., Catterall C.P. & Piper S.D. (2006) Factors affecting the use of reforested sites by reptiles in cleared rainforest landscapes in tropical and subtropical Australia. *Restoration Ecology*, 14, 67–76.
(2) Queheillalt D.M. & Morrison M.L. (2006) Vertebrate use of a restored riparian site: A case study on the central coast of California. *The Journal of Wildlife Management*, 70, 859–866.
(3) Bateman H.L., Chung-MacCoubrey A. & Snell H.L. (2008) Impact of non-native plant removal on lizards in riparian habitats in the southwestern United States. *Restoration Ecology*, 16, 180–190.
(4) Bateman H.L., Chung-MacCoubrey A., Snell H.L. & Finch D.M. (2009) Abundance and species richness of snakes along the Middle Rio Grande riparian forest in New Mexico. *Herpetological Conservation and Biology*, 4, 1–8.
(5) Shoo L.P., Wilson R., Williams Y.M. & Catterall C.P. (2014) Putting it back: Woody debris in young restoration plantings to stimulate return of reptiles. *Ecological Management and Restoration*, 15, 84–87.
(6) Mosher K.R. & Bateman H.L. (2016) The effects of riparian restoration following saltcedar (*Tamarix* spp.) biocontrol on habitat and herpetofauna along a desert stream. *Restoration Ecology*, 24, 71–80.

13.19. Create or restore shrubland

- **One study** evaluated the effects of creating or restoring shrubland on reptile populations. This study was in Mexico[1].

COMMUNITY RESPONSE (1 STUDY)

- **Richness/diversity (1 study):** One replicated, controlled study in Mexico[1] found that areas of restored shrubland had similar reptile and amphibian species richness compared to areas that were not restored.

POPULATION RESPONSE (1 STUDY)

- **Abundance (1 study):** One replicated, controlled study in Mexico[1] found that areas of restored shrubland had a higher abundance of lizards than areas that were not restored.

BEHAVIOUR (0 STUDIES)

Background

Loss of shrubland may be due to a range of factors, including too many grazing animals inhibiting regeneration of shrubs, too few grazing animals or fire suppression leading to reversion to woodland, or invasion by non-native species. Shrubland restoration or creation may benefit reptiles associated with the habitat.

A replicated, controlled study in 2009–2010 in three sites of dry scrub within a wider urban setting in Mexico City, Mexico (1) found that restoring shrubland by planting native species, removing invasive plants and constructing rock piles resulted in similar species richness, but higher abundance of lizards compared to a site with no management. Results were not statistically tested, and the effect of each intervention cannot be separated. Restored sites had a similar number of species (4 species of reptiles and amphibians) as the site with no management (3 species of reptiles and amphibians). Higher numbers of lizards were observed in the two restored sites compared to the site with no management (overall abundances not provided). In 2005–2006, restored sites (0.5 and 0.3 ha) were cleared of rubbish and exotic woody vegetation (dominated by *Eucalyptus camaldulensis*); replanted with native vegetation; and rock piles were constructed (2–3 m diameter and 1.2 m high). The site without management (0.3 ha) had no vegetation removal or planting or rock piles. Sites were surveyed eight times each by slow, random walks between May 2009–2010.

(1) San-José M., Garmendia A. & Cano-Santana Z. (2013) Vertebrate fauna evaluation after habitat restoration in a reserve within Mexico City. *Ecological Restoration,* 31, 249–252.

13.20. Restore beaches

- **One study** evaluated the effects of restoring beaches on reptile populations. This study was in the USA[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Reproductive success (1 study):** One controlled, before-and-after study in the USA[1] found that removing beach debris from one section of beach did not increase nesting success in that section.

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One controlled, before-and-after study in the USA[1] found that after the removal of beach debris from one of three beach sections, a higher percentage of both the total nests laid and failed nesting attempts occurred in that section.

Background

Beach restoration may include vegetation management or removal of debris to maximise nesting site availability and access for reptiles. The process of dune re-mobilization is one commonly used management technique, though data is often insufficient for evaluating the impact of this intervention on reptile populations (e.g. Hill *et al.* 2018).

Studies that discuss physical rebuilding of beaches, for example by bringing sand to replenish a beach ('beach nourishment') are discussed in *Threat: Natural system modifications – Restore or maintain beaches ('beach nourishment').* Studies that discuss beach stabilization to protect against severe weather are discussed in *Threat: Natural system modifications – Armour shorelines to prevent erosion*.

See also: *Restore island ecosystems*.

Hill P., Moulton N. & Foster J. (2018) *Sand lizard surveys at Newborough Warren NNR and sand dune habitat management guidance*. Natural Resources Wales report 302.

A controlled, before-and-after study in 2011–2014 on a beach in north-west Florida, USA (1) found that restoring a beach by removing debris (man-made and natural) increased both the percentage of total loggerhead turtle *Caretta caretta* nests laid and failed nesting attempts in the restored section, and that nesting success remained similar when debris was left in place. The percentage of total nests that were laid in the beach section cleared of debris increased after removal (27 of 84 nests, 32%) compared to before (9 of 74 nests, 12%), whereas the percentage of total nests laid in the two no-removal sections decreased in one case (after: 15%; before: 32%) and stayed the same in the other (after: 52%; before: 58%). The percentage of failed nesting attempts ('false crawls') in the beach section cleared of debris also increased after removal (45 of 131 crawls, 34%) compared to before (29 of 170 crawls, 17%), and decreased in the two no-removal sections (after: 15–50%; before: 25–58%). Nest success rate was similar after debris removal (after: 38% success; before: 24% success). The beach (5.7 km total length) was divided into three sections that initially had natural debris only (1.3 km long); man-made and natural debris (1.7 km long, 'middle'); or comparatively little debris (2.7 km long). All man-made (concrete, pipes, metal fencing) and natural (fallen trees and stumps due to erosion of an adjacent pine forest) debris were recorded (June–December 2012) and removed from the middle section only in December 2012. Nesting activity was monitored on all three beach sections daily in May–September 2011–2014 (two years before and after removal).

(1) Fujisaki I. & Lamont M.M. (2016) The effects of large beach debris on nesting sea turtles. *Journal of Experimental Marine Biology and Ecology*, 482, 33–37.

13.21. Create or restore waterways

- **Two studies** evaluated the effects of creating or restoring waterways on reptile populations. Both studies were in the USA[1,2].

COMMUNITY RESPONSE (1 STUDY)

- **Community composition (1 study):** One site comparison study in the USA[1] found that restored and pristine streams had similar turtle community composition.

POPULATION RESPONSE (2 STUDIES)

- **Abundance (2 studies):** One site comparison study in the USA[1] found that restored and pristine streams had a similar abundance of turtles. One replicated, site comparison study in the USA[2] found that creating new waterways by redirecting flows during forest restoration had mixed effects of reptile abundance.

BEHAVIOUR (0 STUDIES)

Background

Streams may be drained or redirected during the development of agriculture or expansion of urban areas or other land uses. Many waterways have been cleared of submerged wood ('desnagging') for recreational purposes (*e.g.* fishing) or to allow boat access or protect against perceived flooding risks.

River restoration projects often include returning large woody material into streams and rivers ('resnagging'), to slow down river flow, provide shelter for native fishes and increase basking areas for reptiles, especially freshwater turtles (Bodie 2001). Other restoration activities may include planting new or managing existing vegetation to provide appropriate shading levels for reptiles, and redirecting water flows.

For studies discussing wetlands and pond creation and restoration, see *Create or restore wetlands* and *Create or restore ponds*.

Bodie J.R. (2001) Stream and riparian management for freshwater turtles. *Journal of Environmental Management*, 62,443–444.

A site comparison study in 2009 in 12 streams in North Carolina, USA (1) found that restored streams had similar overall turtle abundance and community composition to pristine stream habitats. Turtle abundance was statistically similar in restored streams (9 turtles/site) compared to pristine streams (4 turtles/site). Overall, turtle communities were statistically similar between restored and pristine streams, although turtle community composition was correlated with habitat characteristics (water quality and habitat structure) that were significantly different between restored and pristine streams (see original paper for details, including abundances of the eight turtle species captured). Turtle abundance and species richness was compared in six restored and six natural, undegraded streams by trapping turtles using hoop nets in May–July 2009 (12 total trap nights/site). Restored streams were in their second to fifth growing season after restoration and shared similar characteristics (see original paper for details of restoration approach). Pristine streams were selected based on biological integrity and proximity to restored streams. Captured turtles were weighed, individually marked (or assessed for distinguishing scars) and released.

A replicated, site comparison study in 2012–2014 in saltcedar *Tamarix ramosissima*-cottonwood *Populus fremontii* forest along a river in Utah, Arizona and Nevada, USA (2) found that the impact of redirecting water flows as part of forest restoration, along with mechanical tree removal, herbicide treatment and replanting native species, on overall lizard abundance was mixed. Trapping surveys indicated that overall lizard abundance was similar in restored stands (127–171 lizards/site/100 trap nights) compared to unrestored stands (62–74), whereas visual encounter surveys found that overall reptile abundance was greater at restored sites (results reported as statistical results). See original paper for the effects of restoration on individual species. In winter–spring 2012–2013, restoration of saltcedar-cottonwood/willow *Salix* spp stands was carried out along the Virgin River, including: introducing/redirecting water flows by trenching, mechanically removing 50% of saltcedar and Russian olive *Elaegnus angustifolia*, spraying stumps with herbicide and transplanting native plants. Saltcedar in Utah was subject to biocontrol by northern tamarisk beetles *Diorhabda carinulata* from 2006 (see original paper for details). Reptiles were monitored in two restored and six unrestored stands in May–July 2013–2014 using drift fences with pitfall and funnel traps (1,060 total trap days) and visual encounter surveys (3 transects/site, see original paper for details).

(1) Dudley M.P., Ho M. & Richardson C.J. (2015) Riparian habitat dissimilarities in restored and reference streams are associated with differences in turtle communities in the Southeastern Piedmont. *Wetlands*, 35, 147–157.

(2) Mosher K.R. & Bateman H.L. (2016) The effects of riparian restoration following saltcedar (*Tamarix* spp.) biocontrol on habitat and herpetofauna along a desert stream. *Restoration Ecology*, 24, 71–80.

13.22. Create or restore wetlands

- **Seven studies** evaluated the effects of creating or restoring wetlands on reptile populations. Six studies were in the USA[1,3-7] and one was in Kenya[2].

COMMUNITY RESPONSE (3 STUDIES)

- **Richness/diversity (3 studies):** One before-and-after, site comparison study in the USA[3] found that reptile species richness and diversity tended to be lower in a restored wetland compared to an undisturbed wetland. One replicated, site comparison study in the USA[5] found that created, restored, enhanced and natural wetlands had similar combined reptile and amphibian species richness. One site comparison study in the USA[6] found that created wetlands and adjacent natural forest had similar reptile species richness and diversity.

POPULATION RESPONSE (2 STUDIES)

- **Reproductive success (2 studies):** One site comparison study in the USA[1] found that a created wetland was used by snapping turtles for egg laying. One before-and-after, site comparison study in the USA[3] found that in a restored wetland, 16 snake, six lizard and eight turtle species successfully reproduced.

BEHAVIOUR (4 STUDIES)

- **Use (4 studies):** One site comparison study and three before-and-after studies (including one replicated study) in the USA[1,4,7] and Kenya[2] found that created or restored wetlands were used by black rat snakes and snapping turtles[1], turtles, lizards, green grass snakes and terrapins[2], six[4] or 18[7] reptile species.

Background

Loss and degradation of wetlands are a major factor contributing to the global decline in reptiles (Gibbons *et al*. 2000). Many wetlands have been drained and altered to allow for agricultural and urban development. Creating new or restoring wetlands may replace some of the habitat lost and therefore help to maintain and increase populations of reptile species that rely on wetlands.

Creating wetlands for wildlife may involve excavating holes in the ground and damming streams (Adams & Saenz 2011) and bespoke design and planning of different wetland zones, water depths and nutrient flows (Mitsch *et al*. 1998). Restoring wetlands may involve tree planting, vegetation management to optimise shading and provide basking sites, and/or water flow management.

Studies investigating the creation of individual ponds or waterways are discussed in *Create or restore ponds* and *Create or restore waterways*.

Adams C.K. & Saenz D. (2011) Use of artificial wildlife ponds by reptiles in eastern Texas. *Herpetological Bulletin,* 115, 4–11.

Gibbons J.W., Scott D.E., Ryan T.J., Buhlmann K.A., Tuberville T.D., Metts B.S., Greene J.L., Mills T., Leiden Y., Poppy S. & Winne C.T. (2000) The Global Decline of Reptiles, Déjà Vu Amphibians. *BioScience*, 50, 653–666.

Mitsch W.J., Wu X., Nairn R.W., Weihe P.E., Wang N., Deal R. & Boucher C.E. (1998) Creating and restoring wetlands. *BioScience*, 48, 1019–1030.

A site comparison study in 1995–1996 of two forested wetlands in Maryland, USA (1) found that some reptiles colonised a created forested wetland. Black rat snakes *Elaphe obsoleta* were seen basking and several snapping turtles *Chelydra serpentina* laid eggs in a created forested wetland. A single five-lined skink *Eumeces fasciatus* was trapped in the adjacent natural wetland but none were trapped in the created wetland. As mitigation for loss of wetland, a 9 ha wetland was constructed in 1994, of which 5.5 ha was forested wetland. Reptiles were captured in pitfall and funnel traps along drift-fencing within the created and adjacent natural forested wetland. Trapping was conducted several times in 1995–1996.

A before-and-after study in 1996–1997 of a created wetland in Nairobi, Kenya (2) found that reptiles used the wetland. Turtles, lizards, green grass snakes and terrapins were recorded in the wetland. In 1996, a 0.5 ha wetland was constructed using a combination of a sub-surface horizontal flow system planted with *Typha*, followed by a series of three pond systems planted with a variety of species including local reeds and ornamental plants. Ponds were shallow near the shore with deep sections in the centre (1.5 m).

A before-and-after, site comparison study in 1995–1996 of a degraded forested wetland in South Carolina, USA (3) found that restoration increased numbers of reptile species over the first four years. Twenty-four snake species, nine lizard species, nine turtle species and American alligator *Alligator mississippiensis* were captured in the restoration area. Successful reproduction was documented for 16 snake, six lizard and eight turtle species. It was assumed that there were no reptiles prior to restoration. However, species diversity (in one of three years) and overall richness was lower in the restored compared to natural site (results presented as indices). Planting regimes, burning or herbicide application had little effect on species assemblage. Restoration included tree planting in 1993–1995 (549–1,078 trees/ha). In some areas herbicide application and prescribed burns were undertaken to control scrub. Approximately 25% of the restoration area was left as unmanaged strips for comparison. Reptiles were monitored over 21 months in planted and unplanted areas and in adjacent natural wetland area using coverboards, minnow traps, turtle traps and hand captures.

A before-and-after study in 1992–1994 in a wetland in Florida, USA (4) found that six reptile species used the wetland within the first two years. The reptiles were first observed six months after the wetland was created and in total six reptile species usually associated with wetlands were recorded. Overall species richness continued to increase throughout the study. A 32 ha wetland was created in July 1992. Reptiles were monitored quarterly from July 1992 to August 1994. Counts were undertaken on transect and perimeter walks.

A replicated, site comparison study in 1999–2000 of 17 wetlands in South Dakota, USA (5) found that combined reptile and amphibian species richness was similar between created, restored, enhanced and natural wetlands. There were a similar number of species in created, restored, enhanced and natural wetlands (1–3 species/wetland). A total of 11 reptile and amphibian species were recorded. Four created, four restored, four enhanced and five natural wetlands were sampled. Wetland creation involved either impounding a small stream or excavating a basin. Restoration included plugging drainage ditches or breaking sub-surface drainage tiles. Enhancement included manipulating water levels to increase wetland size or changing vegetation structure. Wetland creation, restoration and enhancement were carried out within the previous 10 years. Monitoring was undertaken using drift-fences with pitfall traps, fish nets and visual surveys around wetland perimeters in spring and autumn in 1999–2000.

A site comparison study in 1995–1996 of a created wetland and adjacent forest in Maryland, USA (6) found that created wetlands had similar reptile richness and diversity to the adjacent natural forest. Reptile richness or diversity were similar between created wetlands (richness: 2–6 species; Simpson's diversity index: 0.2–0.8) and natural forest (richness: 4; diversity: 0.7). Two of 12 total species were recorded in both created wetland and natural forest. Eight of 12 species were recorded in created wetland but not natural forest and two of 12 species were recorded in natural forest but not created wetland. The 52 ha wetland was constructed in four terraces and was surrounded by regenerating forest. Monitoring was undertaken in March–September 1995–1996 using transects, call counts, drift-fencing with pitfall and funnel traps. The adjacent forest was used as a reference site.

A replicated, before-and-after study in 2000–2004 of three constructed wetlands in southern Illinois, USA (7) found that reptiles colonized, and continued to colonize, wetlands over four years of monitoring. A total of 18 species were recorded including seven turtle species (38–66 individuals/wetland), nine snake species (101–129) and two lizard species (0–2). Five additional reptile species were recorded in the second year after wetland creation, two in the third year and four in the fourth year, suggesting ongoing colonization. Wetlands were created on a former vegetable farm in 1999–2000 by enclosing water behind earth dams at the end of valleys. Hardwood tree seedlings were also planted. Wetlands were surveyed in April–June in 2001–2004. Monitoring was undertaken using drift-fencing (four fences/wetland and three fences/adjacent habitat) with funnel traps (4 traps/fence), artificial coverboards (0.7 m^2), visual encounter surveys and baited hoop net traps (one trap/wetland).

(1) Perry M.C., Sibrel C.B. & Gough G.A. (1996) Wetlands mitigation: partnership between an electric power company and a federal wildlife refuge. *Environmental Management*, 20, 933–939.

(2) Nyakang'o J.B. & vanBruggen J.J.A. (1999) Combination of a well-functioning constructed wetland with a pleasing landscape design in Nairobi, Kenya. *Water Science and Technology*, 40, 249–256.

(3) Bowers C.F., Hanlin H.G., Guynn Jr D.C., McLendon J.P. & Davis J.R. (2000) Herpetofaunal and vegetational characterization of a thermally-impacted stream at the beginning of restoration. *Ecological Engineering*, 15, S101–S114.

(4) Kent D.M. & Langston M.A. (2000) Wildlife use of a created wetland in central Florida. *Florida Scientist*, 63, 17–19.

(5) Juni S. & Berry C.R. (2001) A biodiversity assessment of compensatory mitigation wetlands in eastern South Dakota. *Proceedings of the South Dakota Academy of Science*, 80, 185–200.

(6) Toure T.A. & Middendorf G.A. (2002) Colonization of herpetofauna to a created wetland. *Bulletin of the Maryland Herpetological Society*, 38, 99–117.

(7) Palis J.G. (2007) If you build it, they will come: herpetofaunal colonization of constructed wetlands and adjacent terrestrial habitat in the Cache River drainage of southern Illinois. *Transactions of the Illinois State Academy of Science*, 100, 177–189.

14. Species management

Background

Most of the chapters in this book are aimed at minimizing threats, but there are also some interventions which aim specifically to increase population numbers by increasing reproductive rates and by introducing individuals. Such interventions may be used in response to a wide range of threats. This chapter describes interventions that can be used to increase population size by translocating wild reptiles from one area to another; protecting or relocating reptile eggs and nests; putting in place measures to protect adult reptiles; breeding or rearing reptiles in captivity (ex-situ conservation) to release back into the wild; or by enhancing resources available for reptiles through supplementary feeding.

14.1. Legally protect reptile species

- **Six studies** evaluated the effects of legally protecting reptile species on their populations. Two studies were in the Netherlands[1,5] and one was in each of the USA[2], Australia[3], the Seychelles[4] and Cape Verde[6].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (3 STUDIES)

- **Abundance (2 studies):** One of two studies (including one replicated, site comparison study and one before-and-after study) in the USA[2] and Australia[3] found that waterbodies where turtle harvesting was prohibited had a similar abundance of red-eared sliders and Texas spiny softshell turtles compared to unprotected waterbodies. The other study[3] found that following legal protection and harvest regulation, the density of saltwater crocodile populations increased.
- **Reproductive success (1 studies):** One before-and-after study in the Seychelles[4] found that following legal protection of both green turtles and their habitat, nesting activity increased.
- **Condition (2 studies):** Two studies (including one replicated, site comparison study and one before-and-after study) in the USA[2] and Australia[3] found that in areas with legal protection and/or harvest regulation, Texas spiny softshell turtles[2] and saltwater crocodiles[3] were larger than in areas with no protection or before protection began. One study[2] also found that female red-eared sliders were larger, but males were a similar size in protected compared to unprotected waterbodies.

BEHAVIOUR (0 STUDIES)

OTHER (3 studies)

- **Human behaviour change (3 studies):** Two reviews in the Netherlands[1,5] found that despite legislation protecting reptiles and their habitat, only one of four development projects completed their habitat compensation requirements[1] or that compensatory slow worm habitat was not completed in time[5]. Both studies[1,5] also found that monitoring data was not available[1] or that the success of a slow worm mitigation translocation could not be assessed[5]. One replicated, before-and-after study in Cape Verde[6] reported that following legal protections combined with public awareness campaigns, self-reported harvesting, selling and purchasing of sea turtles and turtle products decreased.

Background

Legal protection can be given to species on a national or international scale. Levels of protection vary for species and may include protection against killing, capturing, disturbing or trading, or damaging or destroying breeding sites or resting places. Legal protection may be complete (for example no take) or partial (for example limitations to harvest that are seasonal or location specific).

Depending on the level of protection, individual species protection may mean that habitats are also protected. This means that activities such as development that are likely to affect protected species and their habitat may be against the law and require licences from a government licensing authority.

Other studies that discuss legal protection of reptiles are included in *Threat: Biological resource use – Regulate wildlife harvesting* and *Habitat protection – Protect habitat*. For studies that look at the effect of specific action plans for species recovery see *Develop/implement species recovery plans.*

A review of habitat compensation for protected reptiles in the Netherlands (1) found that legislation was not effective at protecting habitats and reptiles. Only one of four development projects completed habitat compensation measures as set out within legal contracts. Some of the compensation required was provided in two projects (36–68%) and none by one project. Two projects created compensation habitat after destroying habitat, rather than before as required, and the timing was unknown for the remaining two projects. No monitoring data were available from any project. In the Netherlands, reptile species are protected and loss of habitat for these species must be compensated by creating new equivalent habitat. Thirty-one projects required to undertake compensation were selected from government files, of which four had commenced and impacted reptiles. Projects were assessed on the implementation of proposed measures in the approved dispensation contracts and on monitoring data. Field visits were undertaken.

A replicated, site comparison study in 2008–2009 of freshwater sites in Texas, USA (2) found that prohibiting freshwater turtle harvesting in public water bodies did not increase turtle abundance but did increase the size of Texas spiny softshell turtles *Apalone spinifera* and female red-eared sliders *Trachemys scripta* compared to unprotected water bodies. One–two years after protections were introduced, red-eared slider and Texas spiny softshell turtle abundance was similar in protected waterbodies (slider: 0.15; softshell: 0.01 turtles/trap day) compared to unprotected waterbodies (slider: 0.12; softshell: 0.05 turtles/trap day). Both male and female Texas spiny softshells were longer in protected waterbodies (male: 171 mm; female: 352 mm) than unprotected waterbodies (male: 151 mm; female: 276 mm). Female red-eared sliders were longer on average in protected waterbodies (222 mm) than unprotected waterbodies (210 mm), whereas males were not (protected: 161 mm; unprotected: 163 mm). From 2007, commercial harvest of freshwater turtles was prohibited in public waterbodies (‘protected’) but unregulated in private waterbodies (‘unprotected’). Turtles were monitored using baited hoop nets in 12 public and 48 private waterbodies spread across three counties (17–22 sites/county > 1 km apart; 5,245 trap days) in May–June 2008 and May–July 2009. Harvest levels were high in two counties and low in one county (see original paper for details). Turtles were marked and carapace lengths were measured.

A before-and-after study in 1975–2009 in 12 tidal rivers in the Northern Territory, Australia (3) found that after legal protection and regulated harvests, saltwater crocodile *Crocodylus porosus* increased in density and average crocodile size recorded increased over time. After saltwater crocodiles were legally protected and harvests regulated, relative density of non-hatchling crocodiles increased by >three times (2009 estimate: 5.3 crocodiles/km; 1975 estimate: 1.5 crocodiles/km). The proportion of larger crocodiles (>1.8 m in length) increased over time in all rivers (most common size in 2007–2008: 2.7 m long, and in 1978–1979: 1.5 m long). Saltwater crocodiles were legally protected in the Northern Territory in 1971. Harvest of non-hatchling crocodiles was limited to <200/year and commercial fishing was banned on most rivers. A managed egg harvest was introduced in 1984–2009 (see original paper for details). Crocodiles were surveyed in 12 large tidal rivers using standardized approach (spotlight surveys at night by boat) in June–October in 1975–2009 (11–29 survey years/river, 33–138 km long surveys/river, 682 km total survey length). Crocodile size was estimated when possible, and only crocodiles >0.6 m ('non-hatchlings') were reported. Relative non-hatchling crocodile densities were estimated using the sightings data divided by the length of river surveyed.

A before-and-after study in 1968–1976 and 1981–2008 on sandy beaches on an atoll island, Aldabra atoll, Seychelles (4) found that legal protection for green turtles *Chelonia mydas*, followed by protection of the whole island 15 years later, resulted in an increase in nesting activity. Results were not statistically tested, and the effects of species and habitat protection cannot be separated. Overall nesting activity was estimated to be higher 36–40 years after turtle protection began (2004–2008: 28,200 nesting attempts/year) compared to 13–17 years after turtle protection began (1981–1985: 10,900–16,500 nesting attempts/year). Authors also reported that estimates of nesting activity around the time that turtle protection began ranged from sightings of seven females (11-day survey in 1967), to 2,000–3,000 nests/year (surveys during 1968–1970 and 1975–1976). Protection for turtles began in 1968, with the Green Turtle Protection Regulations 1968, and the atoll became a UNESCO World Heritage Site in 1983. In 1981–2008, up to 68 nesting beaches on the atoll were surveyed for turtle tracks and evidence of nesting. Survey effort was variable between different years and beaches, with beaches surveyed 0–37 times/years in 1981–1994, and 4–171 times/month in 1995–2008.

A review in 2011 of compliance with legislation during development projects in the Netherlands (5) found that evidence was not provided to suggest that legislation protected a population of slow worms *Anguis fragilis*. Mitigation translocations of slow worms to a compensatory area began in 2009, but the new habitat was only considered finished in the year following translocations and so slow worms were released into potentially unsuitable habitat. Monitoring before and after translocation was insufficient to determine population numbers or to assess translocation success. In June–September 2009, one hundred and forty-nine slow worms were translocated from a 1.1 ha area of rough grassland to a 2.1 ha compensation area. In the Netherlands, the Flora and Fauna Act protects amphibians. The development project was required by law to provide a compensation area for slow worms and to translocate the species from the development site to that area.

A replicated, before-and-after study in 2011 on Santiago and Boa Vista islands, Cape Verde (6) reported that after implementing legal frameworks to penalise killing and consumption of marine turtles and protect turtle nests on beaches, as well as public awareness raising campaigns, participation in consumption of turtle products, turtle harvesting and, in some locations, selling turtle products, declined. Results were not statistically tested. After national legal protections for marine turtles were introduced, fishers self-reported a decline in turtle harvesting from 61–87% to 17–18% of survey participants between 2002 and 2011. Fish sellers self-reported a decline in selling turtle products from 78% to 22% of participants on Santiago between 2002 and 2011 (on Boa Vista only one seller self-identified as selling turtle products and continued to do so). The general public self-reported a decline in turtle product consumption on both islands (Boa Vista: 28% decline, Santiago: 62). However, the authors also reported a significant increase in commercial use of turtle meat with trade increasing between Boa Vista and Santiago (see original paper). In 2005 and 2010, legal frameworks were put in place to penalize killing and consumption of marine turtles. Turtle nests on beaches were protected by the military and public awareness campaigns were carried out by local and international NGOs. In May–June 2011, interviews were carried out with individuals from Santiago and Boa Vista coastal communities. Survey participants were fishers (Boa Vista: 46 individuals; Santiago: 82), fish sellers (5; 18) and the general public (94, 189).

(1) Bosman W., Schippers T., de Bruin A. & Glorius M. (2011) Compensatie voor amfibieën, reptielen en vissen in de praktijk. *RAVON*, 40, 45–49.
(2) Brown D.J., Farallo V.R., Dixon J.R., Baccus J.T., Simpson T.R. & Forstner M.R.J. (2011) Freshwater turtle conservation in Texas: harvest effects and efficacy of the current management regime. *Journal of Wildlife Management*, 75, 486–494.
(3) Fukuda Y., Webb G., Manolis C., Delaney R., Letnic M., Lindner G. & Whitehead P. (2011) Recovery of saltwater crocodiles following unregulated hunting in tidal rivers of the Northern Territory, Australia. *The Journal of Wildlife Management*, 75, 1253–1266.
(4) Mortimer J.A., Von Brandis R.G., Liljevik A., Chapman R. & Collie J. (2011) Fall and rise of nesting green turtles (Chelonia mydas) at Aldabra Atoll, Seychelles: positive response to four decades of protection (1968–2008). *Chelonian Conservation and Biology*, 10, 165–176.
(5) Spitzen-van der Sluijs A., Bosman W. & de Bruin A. (2011) Is compensation for the loss of nature feasible for reptiles, amphibians and fish? *Pianura*, 27, 120–123.
(6) Hancock J.M., Furtado S., Merino S., Godley B.J. & Nuno A. (2017) Exploring drivers and deterrents of the illegal consumption and trade of marine turtle products in Cape Verde, and implications for conservation planning. *Oryx*, 51, 428–436.

14.2. Develop/implement species recovery plans

- **One study** evaluated the effects of developing/implementing species recovery plans on reptile populations. This study was in Australia[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (0 STUDIES)

OTHER (1 STUDY)

- **Conservation status (1 study):** One controlled, before-and-after, paired study in Australia[1] found that the chance of a species' conservation status improving or being stable was similar for those with a recovery plan (including three reptile species) and those without a plan (including three reptile species).

Background

Species recovery plans are documents that set out the actions for species management. Recovery plans are more likely to be effective when they are tailored to the species at risk, revised regularly to ensure that they are up to date, and incorporate evaluation to enable efficient and effective conservation action (Boersma *et al.* 2001).

Boersma P.D., Kareiva P., Fagan W.F., Clark J.A. & Hoekstra J.M. (2001) How Good Are Endangered Species Recovery Plans? *BioScience*, 51, 643–649.

A controlled, paired species, before-and-after study in 2010 in Australia (1) found that species with a recovery plan (including 3 reptile species) were not more likely to have improved conservation status compared to species without a plan (including 3 reptile species). The chance of the status of a species being stable or improving was similar for species with a recovery plan (66%) and without a plan (62%). The evaluation assessed species status of 56 species (including 3 reptile species: striped legless lizard *Delma impar*, Bellinger River Emydura *Emydura macquarii signata,* Blue Mountain's water skink *Eulamprus leuraensis*) with a recovery plan and 67 threatened species (including 3 reptile species: Flinders Ranges worm-lizard *Aprasia pseudopulchella*, Mary River turtle *Elusor macrurus*, Krefft's tiger snake *Notechis scutatus ater*) without a recovery plan. All species were listed under the Environment Protection Biodiversity Conservation Act and either had an approved single-species plan or were lacking a federal recovery plan.

(1) Bottrill M.C., Walsh J.C., Watson J.E.M., Joseph L.N., Ortega-Argueta A. & Possingham H.P. (2011). Does recovery planning improve the status of threatened species? *Biological Conservation*, 144, 1595–1601.

Translocations

14.3. Translocate adult or juvenile reptiles

Background

Translocations involve the intentional capture, movement and release of wild-caught reptiles into the wild to re-establish a population that has been lost, or to augment an existing population. This can reduce the risk of inbreeding; help safeguard small populations from extinction due to catastrophic events and/or increase the occupied range. Translocations can also be used to move reptiles to areas where threats have been removed, such as islands where invasive predators have been eradicated. However, translocations are typically expensive and may risk spreading pathogens to previously unexposed areas.

Release techniques vary considerably, from 'hard releases' involving the simple release of individuals into the wild, to 'soft releases' that involve a variety of adaptation and acclimatisation techniques before release or post-release feeding and care.

This action includes studies which may combine different release techniques, but studies that explicitly test these different techniques are summarized separately under *Use holding pens or enclosures at release site prior to release of wild reptiles; Use holding pens or enclosures at release site prior to release of captive-bred reptiles* and *Release reptiles into burrows*.

This action includes the translocation of wild juvenile or adult reptiles. Relocations of eggs and nests, releases of captive bred individuals and releases of head-started individuals (reptiles of wild-origin reared in captivity prior to release) are discussed under: *Relocation of eggs and nests* and *Captive breeding, rearing and releases (Ex-situ conservation)*. For studies that release reptiles outside of their native range see *Release reptiles outside of their native range*.

Translocations that are carried out to mitigate against specific threats (for example translocating problem individuals away from a specific area, or translocating individuals away from development areas) are summarized under *Mitigation translocations – Translocate problem reptiles*; *Translocate reptiles away from threats* and *Temporarily move reptiles away from short-term threats*.

Due to the number of studies found, this action has been split by species group, though no studies were found for amphisbaenians.

Sea turtles

- **Two studies** evaluated the effects of translocating sea turtles on their populations. One study was global[1] and one was in Japan[2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Abundance (1 study):** One global review[1] reported that zero of four sea turtle translocations were considered successful.
- **Reproductive success (1 study):** One global review[1] reported that zero of four sea turtle translocations found that breeding occurred.
- **Survival (1 study):** One study in Japan[2] found that at least two of five wild-caught hawksbill turtles survived at least six months following release.

BEHAVIOUR (1 STUDY)

- **Behaviour change (1 study):** One study in Japan[2] found that at least two of five wild-caught hawksbill turtles returned to their point of capture after release.

A review of worldwide translocation programmes for reptiles during 1962–1990 (1) found that at least half of those involving sea turtles were unsuccessful.

Two of four (50%) programmes were considered unsuccessful, and for a further two the result was unknown. In addition, breeding was not observed in three of four programmes, and for the other the result was unknown. The origin of individuals (wild populations or captive-bred) was not described for all programmes. Published and unpublished literature was searched.

A study in 2005–2006 off the coast of an island in southwestern Japan (2) found that translocated hawksbill turtles *Eretmochelys imbricata* that were held in captivity before release tended to return to their point of capture. Five wild-caught turtles (held in captivity for 4 months) were tracked for 2–8 days, and two were recaptured 182–199 days after release at their original point of capture (around 5–15 km from release site). An additional four head-started turtles were tracked for 4–9 days and a fifth turtle was tracked intermittently for 10 months. Five wild turtles were captured and held in captivity for four months in large rearing tanks (2 or 5 kl). Five head-started turtles were raised for 2.5 years after being hatched from eggs collected on the island. All turtles were fitted with radio transmitters and released in April 2005 following 1 h sea-acclimation in an enclosure net (4 × 4 × 5 m). Turtles were tracked using 12 fixed receivers deployed on the ocean floor (18 m deep).

(1) Dodd C.K. Jr & Seigel R.A. (1991) Relocation, repatriation, and translocation of amphibians and reptiles: are they conservation strategies that work? *Herpetologica*, 47, 336–350.

(2) Okuyama J., Shimizu T., Abe O., Yoseda K. & Arai N. (2010) Wild versus head-started hawksbill turtles Eretmochelys imbricata: post-release behavior and feeding adaptions. *Endangered Species Research*, 10, 181–190.

Tortoises, terrapins, side-necked & softshell turtles

- **Twenty-six studies** evaluated the effects of translocating tortoises, terrapins, side-necked & softshell turtles on their populations. Sixteen studies were in the USA[1,6,7,9,12,15-18,20-26], two were Global[2,10] and one was in each of Italy[3], the Seychelles[4], Madagascar[5], Cameroon[8], Egypt[11], China[13], Jordan[14] and France[19].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (24 STUDIES)

- **Abundance (4 studies):** One replicated study[7] and one of two global reviews[2,10] reported that three of eight translocations of turtles resulted in established populations[7] and 15 of 47 reptile translocations resulted in stable or growing populations[10] (review included both wild-caught and captive bred animals). The other review[2] (both wild-caught and captive bred animals) reported that one of five translocation of tortoises and snapping turtles was unsuccessful and four had unknown outcomes. One study in the Seychelles[4] found that 47% of translocated Aldabra giant tortoises were found 12 years after releaser.
- **Reproductive success (5 studies):** One global review[2] reported that successful reproduction was found in three of five translocations of tortoises and snapping turtles (review included both wild-caught and captive bred animals). Two of three studies (including one replicated, randomized study) in the USA[1], Italy[3] and Madagascar[5] reported successful reproduction in translocated populations of gopher tortoises[1] and radiated tortoises[5]. The other study[3] reported no evidence of reproduction for three years following a translocation of European pond turtles. One replicated study in the USA[24]

found that translocated female Agassiz's desert tortoises bred successfully following release, but all hatchlings were sired by resident tortoises, not translocated males.

- **Survival (16 studies):** Two of four controlled studies (including one replicated study) in the USA[9,12,16,25] found that translocated eastern box turtles[9] and Eastern painted turtles[25] had lower survival than resident turtles. The other two studies[12,16] found that translocated desert tortoises[12] and musk turtles[16] had similar survival to that of resident turtles. One replicated study in the USA[23] found that translocated gopher tortoises released into temporary enclosures had similar survival compared to head-started tortoises also released into temporary enclosures over four years. Five of 11 studies (including two replicated, controlled studies) in the USA[1,6,17,18,22], Italy[3], Cameroon[8], Egypt[11], China[13], Jordan[14] and France[19] reported that 69–100% of 3–40 translocated individuals survived for monitoring periods of one month to two years[8,13,14,18,22]. Four studies[1,3,11,19] reported that 19–43% of 15–109 translocated individuals survived for 2–3 years. The other two studies[6,17] reported that 0–1% of 15 and 249 translocated individuals survived for up to a year.
- **Condition (3 studies):** One controlled study in the USA[25] found that translocated Eastern painted turtles lost more body mass than resident turtles. One controlled, before-and-after, replicated study in the USA[15] found that translocated desert tortoises had similar stress levels compared to resident tortoises. One study in the USA[21] found that one translocated Blanding's turtle grew over a two-year period following release.

BEHAVIOUR (9 STUDIES)

- **Use (2 studies):** Two replicated studies (including one controlled study) in the USA[22,26] found that one of 13 translocated gopher tortoises returned to its point of capture[26], and no Agassiz's desert tortoises translocated >5 km returned to their point of capture[22].
- **Behaviour change (7 studies):** Two of six studies (including three replicated, controlled studies) in the USA[9,16,18,20,25] and Jordan[14] found mixed effects on the movement of translocated red-eared slider turtles in spring or autumn[20] and on the movement and home range size of translocated eastern box turtles[9] compared to residents. Two studies[14,16] found that four measures of behaviour of translocated musk turtles[16] and home range size of translocated spur-thighed tortoises[14] were similar to residents. One study[18] found that translocated gopher tortoises moved more and had larger home ranges than resident tortoises. The other study[25] found that after ephemeral ponds dried up, translocated Eastern painted turtles did not move to new ponds whereas resident turtles did. One replicated study in France[19] found that the home ranges of translocated European pond turtles were smaller in the year after release compared to the year they were released.

A replicated, randomized study in 1985–1987 in mixed pine and cabbage palm woodland in Florida, USA (1) found that over a third of translocated gopher tortoises *Gopherus polyphemus* initially kept in holding pens survived at least two years after release and bred in the wild. In total 32 of 75 tortoises survived at least two years after release. One of three recaptured females was gravid and three tortoises less than a year old were captured two years after the original release. In 1985, a total of 75 tortoises were caught using bucket traps and translocated to a county park 25 km away. Tortoises had previously been present in the new location but they were no longer considered to be present at the time of release. Tortoises were individually marked and were randomly allocated to one of four holding pens (56 m^2) for 0–15 days prior to release. An additional 10 tortoises were released in 1986. Tortoises were recaptured in 1986 and 1987. Female tortoises were x-rayed to check for gravidity.

A review of worldwide translocation programmes for reptiles during 1962–1990 (2) found that none of the five translocations involving tortoises and snapping turtles (Chelydridae spp. and Testudinidae spp.) were successful. One of five translocations was unsuccessful (desert tortoise *Xerobates agassizii*) and four of five had unknown outcomes (gopher tortoise *Gopherus polyphemus*, Galápagos giant tortoises *Geochelone elephantopus*, Aldabra giant tortoise *Aldabrachelys gigantea* and alligator snapping turtle *Macrochelys temminckii*). Breeding was noted in three of the programmes (Galápagos giant tortoise, Aldabra giant tortoise, gopher tortoise). The origin of individuals (wild populations or captive-bred) was not described for all programmes. Published and unpublished literature was searched.

A study in 1989–1992 in a freshwater lake in north-west Italy (3) found that some translocated European pond turtles *Emys orbicularis* released initially into a holding pen (with other associated actions) survived at least three years but there was no evidence of breeding in the wild. Twenty-nine of 41 translocated European pond turtles placed initially in a holding pen were observed at least once in the first four months following their release. Three years after release, at least six of 45 turtles were observed near the release site and a further four turtles were regularly seen 4 km away. No breeding activity was recorded. Fourteen turtles dispersed within two days of release and were not seen again. No dead turtles were found. In spring 1989 and spring 1990, forty-five individually marked European pond turtles were released into a temporary holding pen (13 m^2 with an artificial pool of 3 m^2) next to a freshwater lake (157 m long) in a protected area closed to the public (1989: 41 individuals, 1990: 4 individuals). Most turtles were released from the holding pen after three weeks (two turtles escaped). Turtles were fed at the release site for two weeks to minimise dispersal. Turtles were surveyed in June–September 1989 and periodically in 1990–1992 (no details of survey method are provided). No specific survival information for the 1990 releases is provided.

A study in 1990 in shrubland on Curieuse Island, Seychelles (4) found that a population of translocated Aldabra giant tortoises *Aldabrachelys gigantea* was still present on the island 12 years after translocation attempts began. A total of 117 tortoises (73 adult males, 38 adult females and 6 juveniles) were found (0.4 tortoises/ha overall; 2 tortoises/ha in occupied areas) 12 years after the start of translocations. Thirteen nesting sites and 21 clutches of eggs were also found. At least five dismembered shells were discovered, and 9% of adults had peeling and flaking shells. Around 250 tortoises were translocation to Curieuse Island between 1978–1982 (95 in 1978; 78 in 1980; around 80 in 1982). In July–October 1990, exhaustive surveys were conducted across the whole of the island and tortoises numbered with paint.

A study in 1994 in one reserve in southeast Madagascar (5) reported that translocated radiated tortoises *Geochelone radiata* survived and some mated following release. No mortalities were recorded, and numerous mating events were observed up to a maximum of one year following the release. In May 1994, a total of 169 radiated tortoises (107 females and 62 males) were released at the reserve site. Tortoises came from customs seizures by officials on Réunion Island.

A study in 1982–1988 in one large pond in Massachusetts, USA (6) found that no translocated northern redbelly turtle *Pseudemys rubriventris* hatchlings survived, whereas some head-started turtles survived at least 3–13 years. Zero of 15 translocated hatchlings were re-captured. Larger head-started turtles had the highest annual survival in the first year following release (<65 mm: 36%; 66–95 mm: 66%; ≥96 mm: 92%), but annual survival in year 2–3 following release were similar for all sizes (60–100%). In 1982, fifteen hatchlings were translocated immediately after capture from a nearby pond. In 1979–1988, sixty-eight head-started turtles were released into the same location. Extensive trapping was carried out for 10 years following the release of the translocated hatchlings.

A replicated study in 1980–1999 in five upland sites near to New York, USA (7) found that of eight translocations of turtles, at least three resulted in established populations. Three translocations of three species (common snapping turtle *Chelydra serpentina*, eastern painted turtle *Chrysemys picta picta*, eastern box turtle *Terrapene carolina carolina*) resulted in established populations and a further two translocations of two species were likely to have been successful (eastern painted turtle, eastern box turtle) based on persistence of offspring records. The success of the other three (common snapping turtle, eastern mud turtle *Kinosternon subrubrum*, spotted turtle *Clemmys guttata*) could not be assessed because of insufficient data. In 1980–1995, five species of locally caught turtles of different life stages were translocated to one or two sites. At one of the sites, coarse woody debris and some temporary and permanent freshwater ponds were also added. Monitoring involved funnel traps, drift-fences with pitfall traps, artificial coverboards, visual searches and radio-telemetry.

A controlled study in 1995–1997 in a forested, lowland protected area in Southwest Region, Cameroon (8) reported that a translocated forest hinge-back tortoise *Kinixys erosa* and two translocated Home's hinge-back tortoises *Kinixys homeana* survived for at least 63–443 days following release and had similar home range sizes to resident tortoises. Results were not statistically tested. The forest hinge-back survived for at least 372 days, and the Home's hinge-backs for at least 63 and 443 days following release. Home ranges were 3–15 ha for translocated tortoises and 3–48 ha for resident tortoises. In April–June 1995, three wild-caught tortoises (one forest hinge-back and two Home's hinge-backs) were obtained from local collectors and translocated to the study site, and in June–November 1995, six resident tortoises (three forest hinge-backs and three Home's hinge-backs) were found within the study site to monitor. All tortoises were fitted with radio-transmitters, attached to the rear edge of the top shell. Tortoises were located as often as possible in 1995, around twice/month in 1996, and around once/month in 1997. Recorded locations were used to calculate the home range size.

A controlled study in 2004–2005 in a mixed-forest site in North Carolina, USA (9) found that translocated eastern box turtles *Terrapene carolina* had lower survival than resident turtles. Fewer translocated turtles survived at least one year than residents (translocated: 5 of 10; resident: 10 of 10). Translocated turtles moved similar distances overall but had larger home ranges than residents in three of three measures (see paper for details). In May and June 2004, ten female turtles were translocated to randomly selected locations within the release site (1–38 km from their point of capture). Ten resident female turtles were also captured, and all turtles were fitted with radio transmitters (attached to shell). Turtles were radio-tracked every two to three days during active periods (May to October 2004; March to June 2005) and once/week during hibernation periods (October 2004 to March 2005) for one year post-release.

A review of worldwide reptile translocation projects during 1991–2006 (10) found that a third were considered successful with substantial recruitment to the adult population. Of the 47 translocation projects reviewed (39 reptile species), 32% were successful, 28% failed and long-term success was uncertain for the remaining 40%. Projects that translocated animals due to human-wildlife conflicts failed more often (63% of 8 projects) than those for conservation purposes (15% of 38) and those for research purposes (50% of 5). Success was independent of the source of animals (wild, captive, and combination), life-stage translocated, number of animals released and geographic region (see original paper for details). Translocated animals were adults in 75% of cases, juveniles and sub-adults in 64% of cases and eggs in 4% of cases. Wild animals were translocated in 93% of projects. The most common reported cause of failure was homing and migration with the second most common reported cause being insufficient numbers, human collection and food/nutrient limitation all equally reported. Success was defined as evidence of substantial recruitment to the adult population during monitoring over a period at least as long as it takes the species to reach maturity.

A study in 2005–2007 in the Omayed Protectorate, Egypt (11) found that approximately a fifth of translocated Egyptian tortoises *Testudo kleinmanni* survived two years after being released. Two years after Egyptian tortoises were released, 21 of 109 tortoises were still alive. Eight tortoises were found dead during the two years after release. Dead tortoises had similar body mass to live tortoises (data presented as statistical model outputs). In September 2005, a total of 109 tortoises were released into a protected area (700 km^2). The tortoises had been confiscated from illegal pet markets in May 2005 and were held in captivity for several months prior to release. Monitoring was carried out every 3 days (3–4 hours/day) in a 4 km area around the release site in May–October 2007 by searching for tortoises, following tracks and looking under vegetation.

A controlled study in 2008 in desert scrubland in California, USA (12) found that most translocated desert tortoises *Gopherus agassizii* survived at least eight months after release and had similar mortality rates to resident tortoises. Eight–nine months after being released, 268 of 357 translocated desert tortoises were still alive. Mortality rates of translocated desert tortoises (89 of 357 individuals died, 25%) was statistically similar to mortality rates of resident tortoises in the release area (29 of 140 individuals died, 21%) and resident tortoises outside of the release area (28 of 149 individuals died, 19%). In March-April 2008, a total of 571 desert tortoises were translocated from a military reservation to 14 widely separated, unfenced public lands (across a total area of 1,000 km^2). Translocated tortoises (357 individuals), resident tortoises in the release areas (140 individuals), and resident tortoises outside of the release areas (149 individuals) were radio-tracked at least once a month in March–December 2008. Mortality rates are based on the radio-tracked tortoises only.

A study in 2007 in one mountain stream in Hebei Province, China (13) found that most translocated big-headed turtles *Platysternon megacephalum* survived for at least 3–8 months following release. Eleven of 16 turtles survived for at least 3–8 months between their release and the end of the study period. A further five turtles went missing sometime following release, though their transmitters were recovered (data not provided). In April–September 2007, sixteen wild-caught adult turtles (9 males, 7 females) were purchased from a turtle dealer and released in to the wild. They were fitted with radio trackers and monitored on 4–6 consecutive days, twice/month until November 2017.

A replicated study in 2007–2008 in one mixed forest in Jordan (14) found that translocated spur-thighed tortoises *Testudo graeca* survived at least 4–11 months, and most had similar range sizes to resident tortoises. Six of seven translocated tortoises survived for the whole 11-month study period, and one tortoise was lost after four months when the transmitter failed. The average range size of translocated tortoises that were tracked for 11 months (6 ha, excluding for one female who ranged 98 ha) was similar to resident tortoises (6 ha). Seven tortoises were confiscated from tortoise sellers, and in June 2007, they were equipped with radio transmitters and released at one release site. Two resident tortoises were fitted with transmitters in October 2007 and February 2008. Translocated tortoises were released just prior to the aestivation period in August–September. All tortoises were located three times/week for 4–11 months following release.

A replicated, controlled, before-and-after study in 2007–2009 in four sites of desert scrub in California, USA (15) found that translocated desert tortoises *Gopherus agassizii* did not have elevated levels of stress hormone compared to wild tortoises. There were no differences in stress hormone (corticosterone) levels between translocated (males: 4–12 ng/mL; females: 3–12 ng/mL) and resident tortoises within release sites (males: 3–12 ng/mL, females: 2–10 ng/mL) or residents outside release sites (males: 3–10 ng/mL; females: 3–11 ng/mL), but overall stress levels did vary between years (2007: 4ng/mL; 2008: 9 ng/mL; 2009: 6 ng/mL). In March 2008, translocated tortoises (45 tortoises: 18 females, 27 males) were released into four areas (1.6 km^2 each) between 9–30 km from the point of capture. A further 179 tortoises (72 females, 107 males) resident in the translocated area were monitored that had home ranges within or outside of the release sites (numbers of each not provided). Levels of stress hormone were measured by taking monthly blood samples (1,793 blood samples in total) in April–October, 2007–2009 from translocated tortoises (19–43 individuals/year) and residents from within (34–43 individuals/year) and outside the release areas (19–48 individuals/year).

A replicated, controlled study in 2010–2011 in wetlands within an urban park Kentucky, USA (16) found that translocated musk turtles *Sternotherus odoratus* had similar post-release survival and movement as resident turtles. Nine of 10 translocated turtles survived the whole 10–11-month study period, compared to 10 of 10 resident turtles. Movement distance, activity area, number of wetlands used, and number of wetland shifts were also similar for translocated and resident turtles (see paper for details). Resident (7 males, 3 females) and translocated (4 males, 6 females, from sites 6–20 km from the release site) adult turtles were trapped between March–August 2010 using baited hoop nets. Radio transmitters were fitted to their shells, and resident turtles were released at point of capture and translocated turtles were randomly assigned one of four ponds and released. Turtles were tracked on 2–3 days/week during warm months and once/month during cool months until June 2011.

A replicated study in 2007–2008 along a river in southern Oklahoma, USA (17) found that most translocated alligator snapping turtles *Macrochelys temminckii* were not recaptured in the year following release. In the year of release adults were captured on 46 occasions (249 released) and one year following release adults were captured on 3 occasions. In comparison, individuals from a cohort of captive-bred juveniles were recaptured on 5 occasions (16 released) in the year of release and on 18 occasions the year after release (number of individuals not given). Seven turtles were confirmed to have died following release. Eight predated nests were found in 2007, seventeen in 2008, and one intact nest was found in 2008. Adult turtles (249 individuals) were originally wild-caught and were confiscated from a turtle farm and released in groups of 27–62 at seven pools adjacent to the river in April 2007. An additional 16 captive-bred juveniles were release at one location in June 2007. Turtles were recaptured with baited hoop nets in May–August 2007 and 2008.

A replicated, controlled study in 2012 in two sites of dry scrub and mixed, open forest in southern Georgia, USA (18) found that some translocated gopher tortoises *Gopherus polyphemus* survived for at least a year and moved more than resident tortoises. Of translocated tortoises that were tagged, eight of 10 (site 1) and 10 of 11 (site 2) survived at least one year following release. Translocated tortoises moved more than resident tortoises (translocated female: 894 m, translocated male: 1,637 m; resident female: 237 m, resident male: 1,410 m), and for two measures of home range size, translocated tortoises had larger home ranges than residents (method 1: translocated: 0–147 ha; resident: 0–13 ha; method 2: translocated: 1–256 ha; resident: 0–64 ha). Thirty-two adult tortoises were trapped in August and September 2011 and split equally between two release sites. Tortoises were placed in circular enclosures (1 ha) in starter burrows (1 m long) and held for an average of 281 or 290 days. At each site, 10 and 11 translocated tortoises and eight and seven resident tortoises were fitted with radio transmitters. In June 2012, the enclosure fencing was removed and tortoises were located weekly until October 2012, and then every 1–4 months until June 2013.

A replicated study in 2007–2010 in a brackish reed marsh in southern France (19) found that after translocating European pond turtles *Emys orbicularis* some survived for at least 1–2 years after release. Of 15 individuals released in 2008, twelve survived at least one year, and five at least two years. Of 14 released in 2009, eight survived at least one year. The home range of turtles the year after release (6 ha) was smaller than that of turtles in the year of their release (14 ha). In June–July 2007, thirty mature turtles were captured (30–70 km from release site) and placed in an acclimation enclosure at the release site. A group of 15 was released in April 2008 (10 females; 5 males), and a group of 14 in April 2009 (10 females; 4 males). Turtles were fitted with radio transmitters and were location twice/week for two months after release (May–June) and then once/week (July–October and March–September of the following year).

A replicated, controlled study in 2010–2012 in a stream and a wetland complex in Kentucky, USA (20) found that of three releases of translocated red-eared slider turtles *Trachemys scripta elegans*, one population of spring-released sliders moved more and further afield than autumn-released or resident turtles, but that another population of spring-released sliders did not move more than resident turtles. Red-eared sliders released in spring into a stream moved more (total distance travelled: 8.6 km) and further away from the point of release (average distance from release: 1.4 km) than sliders released in autumn (total distance travelled: 3.8 km; average distance from release: 0.6 km) or resident sliders (total distance travelled: 4.6 km; average distance from release: 0.6 km). In a second translocation to a wetland complex, spring-released sliders had similar sized home ranges (6.3 ha) and travelled similar distances in total (4.5 km) compared to resident turtles (home range: 6.0 ha; total distance travelled: 3.8 km). Twenty-three sliders were translocated into a stream after hibernation in spring (March–May 2011 and 2012; 12 individuals) and before hibernation in autumn (October 2011, 11 individuals) from 20 km away and monitored in March 2011–October 2012 alongside 11 resident sliders (captured in May–October 2011). A further 13 sliders (captured May–August 2010) were translocated to a wetland complex and monitored alongside 13 resident sliders (captured April–June 2010) in May 2010–June 2011. All turtles were radio-tracked twice weekly in the activity season and once a month during hibernation.

A study in 2007–2010 in forested wetlands in eastern Massachusetts, USA (21) found that at least one translocated hatchling Blanding's turtle *Emydoidea blandingii* survived two years in the wild. One translocated and directly released Blanding's turtle hatchling was incidentally recaptured two years later and had increased in size from 10g and 38 mm long at time of release (September 2008) to 105 g and 88 mm long on recapture (October 2010). In August 2007–2009, a total of 81 hatchlings from 36 nests at a source location were taken for direct release at a recipient wetlands refuge (reserve size: 880 ha).

A replicated, controlled study in 2009–2010 in a site of desert scrub in California, USA (22) found that translocated Agassiz's desert tortoise *Gopherus agassizii* all survived and did not return to their home range if they were translocated more than 5 km from the capture site. All tortoises survived at least 37 days (40 individuals) or at least seven months (40 individuals). Nine of 47 translocated tortoises returned to their initial capture site (8 of 18 returned from 2 km; 1 of 15 returned from 5 km; 0 of 14 returned from 8 km) between five and 37 days after translocation. Tortoises were initially located and fitted with radio transmitters (80 individuals). In 2009, tortoises were translocated 2 km (10 individuals), 5 km (7 individuals) or 8 km away (6 individuals) from their capture location or released at their point of capture (17 individuals). In 2010, a further group of tortoises were translocated 2 km (8 individuals), 5 km (8 individuals) or 8 km away (8 individuals) from their capture location or released at their point of capture (16 individuals). In September–October 2009, tortoises were radio tracked for 37 days before being returned to their point of capture. In April–October 2010, tortoises were tracked for 186 days. Tortoises were located 2–7 times/week.

A replicated study in 2001–2006 in open mixed pine forest in South Carolina, USA (23) found that just under a third of translocated gopher tortoises *Gopherus* *polyphemus* held in temporary enclosures for six months survived four years after release and that survival rates tended to be similar to head-started tortoises. Results were not statistically tested. In the first four years after release from temporary enclosures, translocated juvenile gopher tortoises had annual survival rates of 57–81%, compared to 53–93% annual survival for head-started juvenile gopher tortoises. Over the same time period, cumulative survivorship was at least 29% for translocated tortoises compared to 38% for head-started tortoises. In August–September 2001, thirty-five juvenile gopher tortoises (ages: 1–9 years) were translocated to an 800 km^2 forest reserve and initially held in small enclosures for six months and provided with artificial starter burrows and food (11–12 juveniles/enclosure, each enclosure 3.5 m diameter) until their release. Thirty-two hatchlings taken from nests at the same donor site as the translocated tortoises were head-started in climate-controlled conditions from September 2001 to June 2002 (see original paper for details) and then released into a 1 ha enclosure with starter burrows until September 2002, when the enclosure was removed. Tortoises were monitored by live trapping in autumn and spring 2002–2006.

A replicated study in 2008–2012 in desert shrubland in California, USA (24) found that some translocated Agassiz's desert tortoises *Gopherus agassizii* survived at least 4 years in the wild and bred, but that all hatchlings tested, including those from resident females, were sired by resident and not translocated male tortoises. Four years after being released, at least 13 translocated female Agassiz's desert tortoises were found to have laid eggs. Of 34 clutches laid by translocated and resident female Agassiz's desert tortoises, none of 35 hatchlings tested were sired by translocated male Agassiz's desert tortoises. In spring 2008, a total of 570 tortoises (184 females, 293 males, 93 juveniles) were translocated to a 1,000 km^2 area, which already had a population of resident tortoises. In April–July 2012, clutches laid by translocated (13 individuals) and resident (21 individuals) female tortoises were located and hatchlings (35 individuals) tested to establish paternity by comparing DNA against a reference database (which included 190 resident males and 305/386 translocated adult and juvenile males). If no significant match with the database was found, the authors assumed that an unknown resident tortoise was the sire. Both translocated (31 individuals) and resident male tortoises (37 individuals) were known to frequent the study area.

A controlled study (year not stated) in a woodland-wetland complex in Maryland, USA (25) found that translocated Eastern painted turtles *Chrysemys picta* had higher mortality than resident tortoises and did not navigate successfully from dry ephemeral ponds to alternative water sources, regardless of season of release. Mortality rates were higher in translocated turtles (early-season release: 10 of 20 turtles; late-season release: 2 of 30 turtles) compared to resident turtles (0 of 60 turtles). After ephemeral ponds dried, no translocated turtles successfully navigated to alternative water sources (within 21 days), although all resident turtles did (within 33 h). Translocated turtles deviated more widely from established turtle navigation routes regardless of release season (early-season release 74 m from route; late-season release: 86 m) compared to resident turtles (1 m). Translocated turtles spent more time stopped, moved slower, took longer to move after stopping and lost more body mass than resident turtles regardless of release season (see original paper for details). The translocation destination habitat included temporary ponds (3 ha each) which dry up within a day each summer. Early-season released turtles (20 individuals) moved 3 months (in April) before draining took place to allow time to learn to navigate the destination habitat before draining occurred, late-season released turtles (30 individuals) moved approximately 96 h before ponds drained (in July) and resident turtles (60 individuals) were monitored by radio telemetry every 15 minutes for 12 h/day, 7 days/week for at least 21 days (see paper for details).

A replicated study in 2016 on grassy roadside verges in east-central Florida, USA (26) found that most gopher tortoises *Gopherus polyphemus* translocated short distances did not attempt to go back to their capture location. Only one of 13 translocated tortoises returned to its capture location (after one day in a single 2,058 m movement). All other translocated tortoises remained under vegetation on or near to the roadside verges where they were released and dug burrows (no data provided, see paper for details). Six tortoises (4 females, 2 males) were captured from inland habitats and seven (2 females, 5 males) from coastal habitat were translocated 2–4 km to a roadside corridor during summer 2016. The tortoises were radio tagged and tracked daily during the summer months (approximately 52 tracking events/tortoise) before recapture and return to their original location.

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Snakes

- **Fourteen studies** evaluated the effects of translocating snakes on their populations. Seven studies were in the USA[3-6,10,12a,12b], two were in Antigua[7,8], two were global[1,9] and one was in each of Canada[2], South Korea[11] and Australia[13].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (12 STUDIES)

- **Abundance (6 studies):** One of two global reviews[1,9] reported that 15 of 47[9] reptile translocations (number of snake species not provided) resulted in stable or growing populations (both wild-caught and captive bred reptiles or snakes). The other review[1] reported that the outcome of one indigo snake reintroduction was unknown. One replicated study in the USA[5] found that five of 13 translocations of snakes resulted in established populations. Three studies in Canada[2] and Antigua[7,8] reported that 3–7 years after translocations of red-sided garter snakes[2] and Antiguan racer snakes[7,8], greater numbers were counted.

- **Reproductive success (3 studies):** One global review[1] reported that breeding occurred in nine of 22 reptile translocations (of both wild-caught and captive bred animals). Two studies in Antigua[7,8] reported successful reproduction in translocated populations of Antiguan racer snakes three years after release.

- **Survival (8 studies):** Three controlled studies in the USA[3,10] and Australia[13] found that translocated timber rattlesnakes[3], northern water snakes[10] and dugite snakes[13] had lower survival than resident snakes, and in one study[10] no translocated snakes survived a year. One controlled study in the USA[12a] found that ratsnakes held in captivity for 1–7 years before release had lower survival compared to snakes held for 7–18 days or resident snakes. Four studies (including one replicated, controlled study) in Canada[2], the USA[4,6] and South Korea[11] reported that 12–45% of snakes survived for three months to eight years after release.
- **Condition (1 study):** One controlled study in the USA[10] found that translocated northern water snakes had similar growth rates compared to resident snakes.

BEHAVIOUR (5 STUDIES)

- **Behaviour change (5 studies):** Three of five controlled studies in the USA[3,4,10,12b,13] found that differences in movement[4,12b,13] and home range size[12b] of translocated and resident hognose snakes[4], ratsnakes[12b] and dugite snakes[13] were mixed. The other two studies[3,10] found that translocated timber rattlesnakes[3] and northern water snakes[10] had larger home ranges than residents. One study[3] also found that translocated timber rattlesnakes had higher daily movements than resident snakes.

A review of worldwide translocation programmes for reptiles during 1962–1990 (1) reported that the outcome of one programme involving indigo snakes *Drymarchon corais* was unknown. The origin of individuals (wild populations or captive-bred) was not described for this programme. Published and unpublished literature was searched.

A study in 1985–1988 at a hibernaculum in a limestone sinkhole in Manitoba, Canada (2) found that around 12% of translocated red-sided garter snakes *Thamnophis sirtalis parietalis* were recaptured after one winter and none were recaptured at the donor site. After one winter, 84 of 720 translocated snakes (12%) were recaptured at the hibernaculum. One translocated snake was recaptured during the following autumn and spring, but sampling was disrupted by commercial snake harvesters. None of the 720 translocated snakes were recaptured at the donor site. The authors reported that three years after the translocation the population consisted of around 800 snakes, 427 if which were new captures. The hibernaculum had been uninhabited since heavy flooding in 1974. In September–October 1985, a total of 720 snakes (451 males, 269 females) were collected at a hibernation site and released at an empty hibernaculum 55 km away. All translocated snakes were individually marked by ventral scale clipping. Snakes were collected periodically at both sites in spring and autumn 1986, spring 1987 and spring 1988 (sampling methods not reported).

A controlled study in 1990–1994 in a site of mixed broadleaf forest in Pennsylvania, USA (3) found that translocated timber rattlesnakes *Crotalus horridus* had higher mortality and moved longer distances than resident rattlesnakes. Translocated rattlesnakes had higher mortality (55% died; 6 of 11 rattlesnakes) than resident rattlesnakes (11%; 2 of 18 rattlesnakes). Translocated snakes had higher daily movements (average daily movement: 55–124 m) compared to resident rattlesnakes (10–37 m), moved more overall in two of three comparisons and had larger ranges in nine of 12 comparisons (see paper for details). In 1991–1992, eleven rattlesnakes (five females, six males) were equipped with radio transmitters and translocated between eight and 172 km from point of capture. In 1990–1992, eighteen resident rattlesnakes (10 females, eight males) were also monitored using radio telemetry. Rattlesnakes were located once every two days for six months of the year (mid-April to mid-October) for up to a year.

A controlled study in 1992–1994 in a site of deciduous forest in Arkansas, USA (4) found that translocating eastern hognose snakes *Heterodon platirhinos* resulted in only one surviving until hibernation. Similar numbers of translocated snakes were predated (5–6 of 8) compared to resident snakes (6 of 8). Translocated snakes survived 3–75 days, whereas residents survived 24–183 days (values taken from table). Average movement of translocated snakes (120 m/day) was similar to residents (119 m/day), but variability in daily movement was nearly six times higher for translocated snakes than residents. Translocated snakes were captured offsite from various localities 8–40 km from the translocation site and resident snakes were captured on site. Snakes were implanted with radio transmitters and released within 5 days of capture in a grassy clearing. Snakes were located daily from April to October 1992–1994.

A replicated study in 1980–1999 in five upland sites near to New York, USA (5) found that of 13 translocations of snakes, at least five resulted in established populations. Five of 13 translocations of three species (northern brown snake *Storeria dekayi*, eastern milk snake *Lampropeltis triangulum triangulum*, black racer *Coluber constrictor*) resulted in established populations; four translocations of four species were likely to have been successful based on persistence of offspring records (smooth green snake *Opheodrys vernalis*, eastern hognose snake *Heterodon platirhinos*, eastern milk snake, black racer) and one translocation of eastern hognose snakes failed. The success of three translocations could not be assessed because of insufficient data (smooth green snake, eastern hognose snake, northern water snake *Nerodia sipedon*). In 1980–1995, six species of locally caught snakes of different life stages were translocated to one or more of five sites. At one of the sites, coarse woody debris and some temporary and permanent freshwater ponds were also added. Monitoring involved funnel traps, drift-fences with pitfall traps, artificial coverboards, visual searches and radio-telemetry.

A study in 1980–2001 on an island off the coast of Florida, USA (6) found that a small number of released eastern indigo snakes *Drymarchon couperi* survived 5–8 years in the wild. In the 17–20 years after 40 eastern indigo snakes were released, five snakes were recorded in the wild and the last snake was observed 5–8 years after release (1983: 1 individual; 1985: 1 individual; 1986: 2 individuals; 1988: 1 individual). In 1980–1982, forty eastern indigo snakes (hatchlings and juveniles from a captive breeding colony, wild-caught adults, confiscated snakes and donated from zoos) were released onto St Vincent Island National Wildlife Refuge (51 km^2). Snakes were monitored using combinations of cameras in gopher tortoise *Gopherus polyphemus* burrows and drift fence/pitfall trap arrays in autumn, winter and spring 1983–1990, January and December 2000, and April 2001. Sightings (unverified) were also recorded but are not reported here.

A study in 1999–2006 on a coastal forest offshore island in Antigua (7) found that a population of Antiguan racer snakes *Alsophis antiguae* translocated to a rat-free island survived at least seven years and bred in the wild. After three years, the first adult offspring was recorded (one individual). After four years, 15 new snakes were recorded. After seven years, the total population was estimated to be 40–50 adult and subadult snakes. Monitoring of the translocated snakes after release indicated that they were hunting and feeding successfully. In total 10 wild-caught snakes were introduced to Rabbit Island (2 ha) in November-December 1999. Five snakes were implanted with radio transmitters for monitoring up to 6 months after their release. All snakes are marked with PIT tags. Black rats *Rattus rattus* were eradicated from the island in 1998.

A study in 2002–2006 on a coastal forest island in Antigua (8) found that a population of Antiguan racer snakes *Alsophis antiguae* translocated to a predator-free island survived at least three years and bred successfully in the wild. Young racer snakes were observed three years after snakes were first translocated and the population had approximately doubled to 98 snakes three years after the first release (data presented in 7). Monitoring of adult female snakes found that two of them had increased in weight and length in the year after being released. In total 45 wild-caught snakes were transported to and released onto Green Island (43 ha) in 2002–2005. Four female snakes were implanted with radio transmitters for post release monitoring in 2003. Black rats *Rattus rattus* were eradicated from the island in 2002 but returned in 2006.

A review of worldwide reptile translocation projects during 1991–2006 (9) found that a third were considered successful with substantial recruitment to the adult population. Of the 47 translocation projects reviewed (39 reptile species), 32% were successful, 28% failed and long-term success was uncertain for the remaining 40%. Projects that translocated animals due to human-wildlife conflicts failed more often (63% of 8 projects) than those for conservation purposes (15% of 38) and those for research purposes (50% of 5). Success was independent of the source of animals (wild, captive, and combination), life-stage translocated, number of animals released and geographic region (see original paper for details). Translocated animals were adults in 75% of cases, juveniles and sub-adults in 64% of cases and eggs in 4% of cases. Wild animals were translocated in 93% of projects. The most common reported cause of failure was homing and migration with the second most common reported cause being insufficient numbers, human collection and food/nutrient limitation all equally reported. Success was defined as evidence of substantial recruitment to the adult population during monitoring over a period at least as long as it takes the species to reach maturity.

A controlled study in 2008–2009 in a site of mixed hardwood forest and scrub patches in Indiana, USA (10) found that translocated northern water snakes *Nerodia sipedon sipedon* had lower survival compared to resident snakes, as well as larger home ranges but similar growth rate. Three of 10 translocated snakes survived five months (annual survival estimated at 20 %) to hibernation but none survived one year compared to seven of 12 resident snakes surviving to hibernation (58% survival) and four surviving to the end of the year (annual survival estimated at 45 %). Translocated snakes had larger home ranges (translocated: 14 ha; resident: 5 ha) but similar growth compared to resident snakes (translocated: 0.07 cm/day & 1.25 g/day; resident: 0.07 cm/day & 0.80

g/day). Translocated snakes were also more likely to leave the release area (40% of snakes) than residents (0%). In May 2008, ten snakes were captured from a site 5 km from the release site, along with 12 resident snakes. All were implanted with radio transmitters (7–11 days in captivity for recovery) before being released. Snakes were located once/week during the active season and once/two weeks when entering and leaving hibernation.

A replicated, controlled study in 2008–2009 in two areas of montane mixed oak forest in central South Korea (11) found that some translocated Amur ratsnakes *Elaphe schrenckii* survived 10 months after release. Five of 11 translocated ratsnakes and two of two resident ratsnakes were alive 10 months after release. Within 10 days of release, three translocated snakes died (two were killed by predators) and two translocated snakes lost transmitter signal. One further translocated snake was lost later in the study. In July 2008, thirteen ratsnakes were surgically implanted with radio-transmitters and released into two valleys in a national park (288 km^2). Two ratsnakes were locally-captured resident females that were released back to their original location (one/valley). Eleven ratsnakes had been illegally collected at least 9 km away from the release sites in April–July 2008 and, after being requisitioned by the police and park rangers, were kept in captivity prior to being translocated (five were released in one valley, six in another). All snakes were radio-tracked July 2008–May 2009: weekly in July–December 2008 and April–May 2009, and monthly in January–March 2009. All translocated snakes that were still being tracked at the end of the study were recaptured and put into a captive breeding programme.

A controlled study in 2012–2015 in a mixed forest and wetland in South Carolina, USA (12a) found that that translocated ratsnakes *Pantherophis obsoletus* that spent longer in captivity had lower survival rates. The longer wild-caught translocated snakes spent in captivity prior to release, the lower their seasonal survival rates were compared to translocated snakes that spent less or no time in captivity (results reported as model outputs). Overall, 11 of 19 wild-caught snakes kept in captivity died after release compared to none of five released directly into the wild and three of 11 wild resident snakes (result was not statistically tested). Snakes that spent more time in captivity were more likely to be found in exposed locations than resident or direct-to-wild translocated snakes (see paper for details). Wild-caught snakes (19 individuals) were kept in captivity for between 13 months and 7 years before being fitted with radio transmitters and released in May 2014. An additional 11 resident snakes and five snakes to be translocated directly from the wild were caught, fitted with transmitters and released into the study site 7–18 days after capture. Snakes were radio tracked five-times/week in May–September 2014, once a week in September–December 2014 and once–twice in spring 2015.

A controlled study in 2012–2015 in a mixed forest and wetland in South Carolina, USA (12b) found that translocated ratsnakes *Pantherophis obsoletus* that spent time in captivity before release had similar home ranges to resident snakes regardless of whether their environment was enriched prior to release, whereas translocated snakes kept in captivity without enrichment moved less each day than snakes translocated directly to the wild. Enriched- and unenriched-captive held snakes had similar home range sizes (enriched: 39 ha; unenriched: 23 ha) to resident snakes (26 ha), whereas snakes translocated directly to the wild had larger home ranges (93 ha). Enriched-captive snakes moved similar average daily distances to resident and direct translocated snakes in June–October, but unenriched-captive snakes moved less than directly translocated snakes in July, August and October (see original paper for details). Wild-caught snakes were kept either in unenriched (10 individuals, individually housed with bedding, food and water) or enriched conditions (9 individuals, additionally provided materials to encourage climbing, foraging and thermoregulatory behaviours) for 13 months to 7 years prior to release. Snakes were fitted with radio transmitters and released in May 2014. In addition, 11 resident snakes and five wild translocated snakes were caught, fitted with transmitters and released into the study site 7–18 days after capture. Snakes were radio tracked > once/week in May–September 2014, once/week in September–December 2014 and once or twice in spring 2015.

A controlled study in 2015–2017 in a suburban area in Perth, Australia (13) found that translocated dugite snakes *Pseudonaja affinis* (urban or problem individuals) had higher mortality rates and larger activity range than resident snakes. Translocated snakes had larger maximum activity ranges (11 m^2/day) compared to resident snakes (1 m^2/day). Translocated snakes travelled similar distances (31 m/day) to resident snakes (11 m/day). All translocated snakes died during the study (4 of 4 individuals) compared to half of the resident snakes (3 of 6 individuals). Deaths were caused by predation or road collisions. In total 10 snakes (four translocated snakes and six resident snakes) were tracked for up to 2 months each in September-December 2015–2017. Snakes were either caught opportunistically in urban areas (two individuals) or reported as problem individuals (eight individuals).

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Lizards

- **Seventeen studies** evaluated the effects of translocating lizards on their populations. Six studies were in New Zealand[2,5,8,11,12,15], three were in the Bahamas[4,13,14a], two were in Australia[9,10], two were global[1,7] and one was in each of the Caribean[14b], St. Lucia[3], Turks and Caicos Islands[6] and Anguilla[16].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (16 STUDIES)

- **Abundance (10 studies):** Three of four reviews that were global[1,7] and in New Zealand[15] and the Caribean[14b] reported that 13–32% of reptile[7] or lizard[1,15] translocations resulted in stable or growing populations (both wild-caught and captive bred animals). The other review[14b] reported that populations from eight of 13 iguana translocations survived for at least 5–20 years. Two of six studies (include one site comparison study) in St. Lucia[3], the Bahamas[4,14a] and New Zealand[5,11,12] reported that translocated lizard populations increased over 3–10 years[3,4]. Two studies[5,11] reported that translocated populations remained stable for one[11] and 6–12[5] years. One study[12] reported that a translocated population declined over 1–2 years. The other study[14a] reported that a translocated population of iguanas survived for at least 40 years.
- **Reproductive success (5 studies):** Two reviews that were global[1] and in New Zealand[15] reported that breeding occurred in 20%[1] and at least 30%[15] of lizard translocations (both wild-caught and captive bred animals). Three studies (including one replicated study) in New Zealand[2], Turks and Caicos Islands[6] and the Bahamas[13] reported successful reproduction in a translocated Whitaker's skink[2] population, a Turks and Caicos Rock Iguana population[6] and one of two San Salvador rock iguana populations[13] after 14 months to five years.
- **Survival (10 studies):** Seven of eight studies (including one replicated, controlled study) in New Zealand[2,5,8,11], Turks and Caicos Islands[6], Australia[9], the Bahamas[13] and Anguilla[16] found that 40–85% of translocated lizards survived for at least 3 months to seven years[2,5,6,9,11,13] or that no mortality was reported in the first year after release[8]. The other study[16] reported that at least one lesser Antillean iguana survived for at least two years. One review in New Zealand[15] found that 9% of lizard translocations (both wild-caught and captive-bred animals) resulted in complete failure (no individuals survived). One site comparison study in New Zealand[12] found that 1–2 years after a translocation of shore skinks, individuals representing three of four pattern types originally released still survived.
- **Condition (1 study):** One replicated, controlled study in Australia[9] found that 67% of Napoleon's skinks gained weight following release.

BEHAVIOUR (2 STUDIES)

- **Use (1 study):** One replicated, controlled study in Australia[9] found that all six Napoleon's skinks translocated to restored mining sites moved into unmined forest within a week of release.

- **Behaviour change (1 study):** One replicated, before-and-after, controlled study in Australia[10] found that provision of artificial burrows and supplementary food affected the use of bare ground areas by pygmy blue tongue lizard translocated into enclosures.

A review of worldwide translocation programmes for reptiles during 1962–1990 (1) found that one of eight lizard translocations were considered successful by providing evidence that a stable breeding population had been established. One translocation of one species was successful (sand lizards *Lacerta agilis*), two translocations of two species were unsuccessful (sand lizards, Saint Croix ground lizard *Ameiva polops*) and four translocations of four species had unknown outcomes (giant girdled lizard *Cordylus giganteus*, Galápagos land iguana *Conolophus subcristatus*, Anegada ground iguana *Cyclura pinguis*, sand lizards). Breeding was noted in two translocations of two species (Galapagos land iguana and sand lizards). The origin of individuals (wild populations or captive-bred) was not described for all programmes. Published and unpublished literature was searched.

A study in 1987–1993 on two islands near North Island, New Zealand (2) found that translocating Whitaker's skinks *Oligosoma whitakeri* to an island following removal of Pacific rats *Rattus exulans* and European rabbits *Oryctolagus cuniculus* resulted in some individuals surviving for at least five years and reproducing. Over a five-year period, 15 of 28 skinks (54%) were recaptured at least once, along with five offspring of translocated skinks. Trapping success ranged from 0.4 skinks/100 trap days (2.5 year after release) to 3.1 skinks/100 trap days (5 years after release, but larger area trapped). On another predator free island 25% of marked skinks were recaptured and trapping success was 0.3 and 0.9 skinks/100 trap days. In 1987, twenty-eight skinks (15 adults, 3 gravid females) were captured using pitfall traps on a predator-free island and translocated to the release island from February 1988 to March 1990. Each skink was released into an artificial burrow and stacks of plywood were provided as extra cover. Pitfall traps were placed in the release are (initially 49, increased to 69 traps over 580 m^2) and were monitored twice/year.

A study in 1995–1998 on a mixed woodland, shrub and grassed island off the east coast of St. Lucia (3) found that a population of translocated St. Lucia whiptail lizards *Cnemidophorus vanzoi* survived at least three years after release and bred. Three years after translocation, the average size of a population of St. Lucia whiptail lizards was estimated to be 145 lizards, more than three times greater than the number of lizards originally released. In 1995, forty-two whiptail lizards taken from a nearby island were released on Praslin Island (1.1 ha). Lizards were surveyed in October–December 1997 and January–March 1998 along line transects and caught with a noose. Black rats *Rattus rattus* were eradicated from the island in 1993 but subsequently encountered there infrequently from 1995 onwards and were removed when discovered.

A before-and-after study in 1988–1998 on a tropical island in the Bahamas (4) found that a translocated population of Allen's Cay iguana *Cyclura cychlura inornata* had grown over a 10-year period following release. Seven of eight iguanas from the original release were recaptured, as well as 28 descendants (11 males, 16 females). The total population was estimated at 40–107 individuals. In

1988–1990, eight iguanas (4 males and 4 females) were translocated to the island from a nearby population. In 1998, six days (2 each in March, May and November) of trapping were carried out and individuals were marked by clipping toes and painting numbers on both sides of the ribs using white correction fluid. Population size was then estimated by walking a 320 m linear transect three times every day for 24 days between March and November 1998 and noting all marked and unmarked individuals.

A study in 1987–1998 on a partially forested island in New Zealand (5) found that after translocating three species of lizard, some individuals were still present 6–12 years later. Up to 38% of the released lizards disappeared in the first 12 months. Lizards had high annual survival *(Cyclodina alani* annual survival rate: 81% male, 88% female; Whitaker's skink *Cyclodina whitakeri*: 76% male, 77% female; egg-laying skink *Oligosoma suteri*: 87% male, 93% female), but adult populations did not increase (*Cyclodina alani*: 4 released, 6 captured after 7 years; Whitaker's skink: 18 released, 11 captured after 12 years; egg-laying skink: 30 released, 35 captured after 6 years). In 1987–1990, lizards (total released: 14, 28 and 30 lizards) were captured in pitfall traps from two islands and translocated to a nearby island where European rabbits *Oryctolagus cuniculus* and Pacific rats *Rattus exulans* had been removed. Lizards were translocated within 48 h of capture, except for 23 Whitaker's skink which were not released for two months. Post-release monitoring was conducted with baited pitfall traps set in spring (November or early December) and late summer (late February or March) and checked daily during a 3–7-day monitoring period.

A study in 1999–2001 on an island in Turks and Caicos (6) found that after the eradication of feral cats *Felis catus,* all translocated Turks and Caicos Rock Iguanas *Cyclura carinata* survived at least two months post release and that some were breeding one year later. All of the first group of translocated iguanas (25 individuals) survived at least two months and 10 individuals survived at least 3–4 months after being released. After 14 months, two hatchlings were observed on the island. An initial group of 25 iguanas was translocated from Big Ambergis to Long Cay island (111 ha) in November 1999. Subsequent translocations took place every 2–3 months thereafter (404 individuals translocated in total). All iguanas were individually marked with PIT-tags. Ten individuals per translocation were fitted with radio collars and radio tracked until the next translocation. Feral cats were removed from the island prior to iguana reintroduction in July 1999 using fish laced with 1080 poison bait (22% concentration, sodium monofluoroacetate) distributed at bait stations placed systematically across the island (every 25 m in parallel lines 50–100 m apart, 500 total bait stations). There were no signs of cats in November 1999. One cat was found in January 2000 and removed from the island.

A review of worldwide reptile translocation projects during 1991–2006 (7) found that a third were considered successful with substantial recruitment to the adult population. Of the 47 translocation projects reviewed (39 reptile species), 32% were successful, 28% failed and long-term success was uncertain for the remaining 40%. Projects that translocated animals due to human-wildlife conflicts failed more often (63% of 8 projects) than those for conservation purposes (15% of 38) and those for research purposes (50% of 5). Success was independent of the source of animals (wild, captive, and combination), life-stage translocated, number of animals released and geographic region (see original paper for details). Translocated animals were adults in 75% of cases, juveniles and sub-adults in 64% of cases and eggs in 4% of cases. Wild animals were translocated in 93% of projects. The most common reported cause of failure was homing and migration with the second most common reported cause being insufficient numbers, human collection and food/nutrient limitation all equally reported. Success was defined as evidence of substantial recruitment to the adult population during monitoring over a period at least as long as it takes the species to reach maturity.

A replicated study on two island off the north-eastern coast of North Island, New Zealand (8) reported that translocated Duvaucel's geckos *Hoplodactylus duvaucelii* survived for at least a year after release and successfully bred. Authors reported that no mortalities were recorded in the first year after release and all recaptured individuals had improved body condition (no data provided). Offspring of the gravid released geckos were recorded after 12 months, and offspring from breeding events on the islands was recorded in 2012. A total of 39 wild geckos were captured on Korapuki Island (50:50 sex ratio) and quarantined for two weeks to test for diseases. All animals were tagged with PIT tags, and 20 geckos were fitted with radio-transmitters. Geckos were released in December 2006; with 19 released on Tiritiri Matangi (220 ha) and 20 on Motuora Island (80 ha). Geckos were monitored intensively in the year after release via a range of methods (including radio-tracking, spotlight surveys and checking funnel traps and artificial refuges), and annual monitoring was conducted thereafter.

A replicated, controlled study in 2008–2009 in eucalypt forest in Western Australia, Australia (9) found that 10 of 12 translocated Napoleon's skinks *Egernia napoleonis* survived at least four months, but all skinks released in restored mine sites moved to unmined forest within a week of being released. Ten of 12 translocated Napoleon's skinks survived at least four months after translocation. One skink lost significant weight, was returned to its source site and was classed as a reintroduction failure. The fate of one skink was unknown as it lost its radio transmitter (the authors reported possibly due to a predatory attack). Of the 10 skinks that remained in the reintroduction sites, eight gained weight after release and two lost a small amount of weight (<1 g). Six of six Napoleon's skinks translocated to restored mine sites moved into unmined forest within 7 days and settled in unmined forest after four months. In November 2008, twelve Napoleon's skinks were released in three 5-year-old restored forest sites and three unmined forest sites (two skinks/site; see original paper for details of restoration). Skinks were radio-tracked weekly for the first four weeks after release and then monthly for the next three months. Skinks were recaptured and weighed monthly.

A replicated, controlled, before-and-after study in 2009 in grass, bare ground and tilled soil enclosures in southern Australia (10) found that translocated Pygmy bluetongue lizards *Tiliqua adelaidensis* provided with artificial burrows and supplementary food were observed less often in bare ground habitat without artificial burrows than lizards that had access to artificial burrows but were not fed. Fed lizards were observed less frequently in bare ground habitat without artificial burrows on most days compared to lizards with the same access to artificial burrows but that were not fed and this effect became larger towards the end of the feeding period (see original paper for details). Fed lizards changed burrow less frequently (0.5 changes/day) compared to unfed lizards (1.1

changes/day). In total 16 lizards were captured and moved to a trial site in a zoo. Four lizards were released into four 15 m enclosed cages in November 2009. Cages contained short grass, bare ground and tilled soil. Artificial burrows were built from hollowed wooden poles (30 cm long, 3 cm diameter) pushed into grassy or tilled soil (82 burrows/cage). No burrows were present in the bare ground habitat. Lizards were fed mealworms daily in burrows for seven days in two cages and not fed in the other two cages, then no lizards were fed for two days before the feeding regime began again, but this time the previously unfed cages were fed daily for seven days and the other cages were not. Lizards were monitored by four surveillance cameras/cage during daylight hours from the second to seventh days of the feeding regime (12 days total).

A study in 2008–2010 in rock outcrops in mixed grass and shrublands in Otago, New Zealand (11) found that most translocated grand skinks *Oligosoma grande* survived the first year and bred in the wild. After one year, 10 of 10 juvenile skinks and five of nine adult skinks had survived. The population increased from 19 to 20 individuals. The authors reported that although the population had increased overall, it's reproductive potential had declined due to the loss of adult skinks. In October 2009, nineteen grand skinks were moved 4 km from three source sites to a cluster of rock outcrops (0.25 ha) in a conservation reserve, where non-native predators had been controlled since 2008. Skinks were monitored every 7–15 days for the first 60 days and in December 2009, April 2010 and December 2010 using photographic surveys. Predators were controlled using traps (800 traps across 4,500 ha of the reserve).

A site comparison study in 2006–2008 on a sand and rock beach on an island in Hauraki Gulf, New Zealand (12) found that a translocated population of shore skinks *Oligosoma smithi* survived at least two years, but that colour pattern variation reduced from four to three pattern types. One–two years after 40 skinks were reintroduced, 29 shore skinks were captured. Four colour pattern types were present in the donor population and the originally translocated skinks, but after 1–2 years, only three colour types were present in the translocated population (see original paper for details). The authors reported that this may have been due to the darker and more vegetated habitats prevalent in the destination location. Forty shore skinks (including nine gravid females) were translocated from a coastal sand dune system and reintroduced to a non-native-predator-free island reserve (220 ha) in 2006. Shore skinks were considered extinct on the island prior to this release. Skinks were released on a dark sand beach with rocks and boulders on the east of the beach and low-level vegetation away from the shoreline. In February 2007–March 2008 skinks were monitored every three months at the source and destination location using 2–3 baited pitfall trapping grids for 3–6 trap nights at a time (see original paper for details). Skinks were photographed, assessed against the habitat background and individually marked prior to release.

A replicated study in 2000–2013 on an offshore cay in San Salvador, Bahamas (13) found that two translocated populations of San Salvador rock iguanas *Cyclura rileyi rileyi*, survived at least seven years in the wild but there was only evidence of breeding in one population. At least two years after a first translocation, five marked San Salvador rock iguanas and one subadult were observed (confirming breeding had taken place in the wild) and three unmarked iguanas were trapped;

after five more years, two adult iguanas were observed. The authors reported that no iguanas from this translocation had survived after 12 years. Seven years after a second translocation, 12 of 14 adult iguanas were still alive, but there was no evidence of breeding in the wild. The first translocation was unsanctioned and took place in November 2000 (or earlier) with at least five individually-marked iguanas translocated to a private resort. The authors reported that feral cats *Felis catus*, dogs *Canis lupus familiaris* and rats *Rattus rattus* were removed from the resort. The authors observed and trapped these iguanas in October 2002, June 2007 and interviewed resort staff about them in 2012. In February 2005 a second translocation of 14 adult iguanas from a neighbouring island took place. These iguanas were surveyed in June 2006 and 2007, January and May 2012, and June 2013.

A study in 1973 and 2002–2013 in sandy palm forest, scrub and rock cay in the Exuma Islands, south-eastern Bahamas (14a) found that a population of translocated Acklins rock iguanas *Cyclura rileyi nuchalis* survived at least 40 years in the wild and bred, although there was some evidence of a population decline in the last year of the study. A population of translocated Acklins rock iguanas fluctuated between an estimated 59–322 individuals 30–38 years after the founding animals were released. Between the 37th and 38th year after release, the population estimate reduced from 218 individuals to 59 individuals. The authors reported that this decline may have been the result of rat predation or poaching. The authors reported that in 1973, five Acklins rock iguana were released on a cay (3.3 ha) in the Exuma Cays Land and Sea Park. Iguanas were monitored by live catching or trapping during daylight hours in May 2002–2005 and May or June 2007–2013.

A review of studies in The Bahamas, Turks and Caicos Islands, Puerto Rico, Grand Cayman and British Virgin Islands (14b) reporting on known translocations of rock iguanas (*Cyclura*) found that eight of 13 translocated populations survived at least 5–20 years in the wild and four of 13 translocations were deemed as being successful or the population had established. Three translocated populations of rock iguanas survived at least 6–10 years, one population survived at least 11–15 years, one population survived at least 16–20 years, and three populations survived more than 20 years in the wild. One population was described as 'reproducing', another as 'established' and two more as 'successful'. The outcome of one translocation was unknown. Between the 1960s and 2012, thirteen populations of eight different rock iguana species (*Cyclura cychlura inornate, Cyclura cychlura figginsi, Cyclura rileyi nuchalis, Cyclura rileyi rileyi, Cyclura carinata, Cyclura nubila nubila, Cyclura lewisi* and *Cyclura pinguis*) were translocated to different island locations. Founder populations ranged from "a few" to 800 individuals and one population comprised headstarted individuals (see original paper for details).

A review published in 2016 of lizard translocation projects in New Zealand during 1988–2013 (15) found that most projects found evidence of breeding following release, but few found evidence of population growth. Forty-five of 53 (85%) translocations motivated by conservation had some post-release monitoring. Seven found evidence of population growth (more lizards found than released), 33 found that populations were smaller than the number released, at least 16 found evidence of breeding after release, and five resulted in complete failure (no lizards found). One translocation (of speckled skinks *Oligosoma infrapunctatum*) was later discovered to be at a location outside the species historic range. Some translocations involved wild animals and others captive bred (project success vs source of animals not stated). Published and unpublished literature were searched, and key people associated with each translocation were identified and contacted for further information. Translocations were considered to be motivated by conservation if the primary focus was to benefit the species or recipient site.

A study in 2016–2018 on a tropical island off the coast of Anguilla (16) reported that at least one translocated lesser Antillean iguana *Iguana delicatissima* survived for at least two years after release. Eight iguanas survived for at least nine weeks after release, and authors reported that individuals continued to be resighted after that point, with one individual being recaptured two years after release. In 2016, a total of 11 iguanas were translocated from the mainland (Anguilla) to a nearby small island (Prickly Pear East; 32 ha). Eight iguanas were fitted with radio collars and relocated every week for nine weeks. Collars were then removed and monitoring was carried out on an ad hoc basis.

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(16) ANT/ATE/STENAPA (2018) *Lesser Antillean Iguana Iguana delicatissima Conservation Strategy and Action Plan for the Northern Caribbean Sub-region (Anguilla, St. Barthélemy, St. Eustatius), 2018–2023*. Anguilla National Trust, Agence Territoriale de l'Environnement and St. Eustatius National Parks Foundation.

Crocodilians

- **Two studies** evaluated the effects of translocating crocodilians on their populations. Both studies were global[1,2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Abundance (2 studies):** Two global reviews[1,2] reported four of five[1] crocodilian translocations and 15 of 47[2] reptile translocations resulted in stable or growing populations (included both wild-caught and captive bred animals).
- **Reproductive success (2 studies):** One global review[1] reported that breeding occurred in at least two of five crocodilian translocations (included both wild-caught and captive bred animals).

BEHAVIOUR (0 STUDIES)

A review of worldwide translocation programmes for reptiles during 1962–1990 (1) found that four of five translocations of crocodilians were considered successful by providing evidence that a stable breeding population had been established. Four translocations of four species were considered successful (American alligator *Alligator mississippiensis*, mugger *Crocodylus palustris*, saltwater crocodile *Crocodylus porosus*, and gharial *Gavialis gangeticus*) and the success of the other translocation was unknown (Nile crocodile *Crocodylus niloticus*). Breeding was noted in two of the translocation programmes (American alligator and gharial). The origin of individuals (wild populations or captive-bred) was not described for all programmes. Published and unpublished literature was searched.

A review of worldwide reptile translocation projects during 1991–2006 (2) found that a third were considered successful with substantial recruitment to the adult population. Of the 47 translocation projects reviewed (39 reptile species), 32% were successful, 28% failed and long-term success was uncertain for the remaining 40%. Projects that translocated animals due to human-wildlife conflicts failed more often (63% of 8 projects) than those for conservation purposes (15% of 38) and those for research purposes (50% of 5). Success was independent of the source of animals (wild, captive, and combination), life-stage translocated, number of animals released and geographic region (see original paper for details). Translocated animals were adults in 75% of cases, juveniles and sub-adults in

64% of cases and eggs in 4% of cases. Wild animals were translocated in 93% of projects. The most common reported cause of failure was homing and migration with the second most common reported cause being insufficient numbers, human collection and food/nutrient limitation all equally reported. Success was defined as evidence of substantial recruitment to the adult population during monitoring over a period at least as long as it takes the species to reach maturity.

(1) Dodd C.K. Jr & Seigel R.A. (1991) Relocation, repatriation, and translocation of amphibians and reptiles: are they conservation strategies that work? *Herpetologica*, 47, 336–350.

(2) Germano J.M. & Bishop P.J. (2009) Suitability of amphibians and reptiles for translocation. *Conservation Biology*, 23, 7–15.

Tuatara

- **Four studies** evaluated the effects of translocating tuatara on their populations. Three studies were in New Zealand[1,2,4] and one was global[3].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (4 STUDIES)

- **Abundance (2 studies):** One global review[3] reported that 15 of 47 reptile translocations resulted in stable or growing populations (review included both wild-caught and captive bred animals). One study in New Zealand[2] found that nine years after a translocation of 32 tuatara to an island where they had previously gone extinct, there was a population of 50 individuals.
- **Reproductive success (2 studies):** One of two studies (including one controlled study) in New Zealand[1,2] reported successful reproduction in one population of translocated tuatara[2]. The other study[1] reported no breeding during the six years following translocation.
- **Survival (2 studies):** Two studies (including one controlled study) in New Zealand[1,4] reported that 61–73% of translocated tuatara were recaptured over a six year period[1] or survived for 9–12 month following release[4].
- **Condition (1 study):** One controlled study in New Zealand[1] found that translocated adult tuatara increased their body weight by 41% following release.

BEHAVIOUR (0 STUDIES)

A controlled study in 1995–2000 on an island in New Zealand (1) found that most translocated tuatara *Sphenodon guntheri* survived at least five years following release but did successfully breed. Eleven of 18 adults (61%) were recaptured over six years following release, as well as 28 of 50 head-started juveniles (56%). Following translocation, adults increased in weight by 41%, and two years after translocation they were heavier than equivalent length individuals from the founder population. No successful breeding was observed during the six-year period, though tuatara are an extremely long-lived species (up to 100 years). In November 1995, eighteen adults (11 females, 7 males) were translocated from North Brother Island to Titi Island (a rodent free island), along with 50 head-started juveniles. Tuatara were released into artificial burrows at night (2,100–2,230 h). Six post-release monitoring trips were conducted between November

1995 and November 2000, when a team of 3–4 people spent up to seven nights on the island searching for tuatara.

A study in 1996–2005 on an offshore island in New Zealand (2) found that a population of tuatara *Sphenodon punctatus* translocated to an island where invasive species had been eradicated survived at least nine years and bred. Numbers of tuatara were estimated to be approximately 50 individuals nine years after they were first released. At least two separate clutches of offspring (indicated by several different sized juveniles) were observed on the island. In 1996, thirty-two adult tuatara were translocated to Motuhora (Whale Island; 143 ha). European rabbits *Oryctolagus cuniculus* and black rats *Rattus rattus* were eradicated from the island using poison bait (Bromadialone, Brodifacoum and 1080 poison) and lethal traps in 1985–1987 (see original paper for details).

A review of worldwide reptile translocation projects during 1991–2006 (3) found that a third were considered successful with substantial recruitment to the adult population. Of the 47 translocation projects reviewed (39 reptile species), 32% were successful, 28% failed and long-term success was uncertain for the remaining 40%. Projects that translocated animals due to human-wildlife conflicts failed more often (63% of 8 projects) than those for conservation purposes (15% of 38) and those for research purposes (50% of 5). Success was independent of the source of animals (wild, captive, and combination), life-stage translocated, number of animals released and geographic region (see original paper for details). Translocated animals were adults in 75% of cases, juveniles and sub-adults in 64% of cases and eggs in 4% of cases. Wild animals were translocated in 93% of projects. The most common reported cause of failure was homing and migration with the second most common reported cause being insufficient numbers, human collection and food/nutrient limitation all equally reported. Success was defined as evidence of substantial recruitment to the adult population during monitoring over a period at least as long as it takes the species to reach maturity.

A study in 2012–2013 in regenerating temperate forest in South Island, New Zealand (4) found that most translocated and captive-reared tuatara *Sphenodon punctatus* survived at least 9 months in the wild. After 3–6 months, all translocated and almost all captive-reared and released tuatara survived (translocated: 100%, captive-reared: 96–100% survival rate). After 9–12 months, survival rates of translocated tuatara (73%) were highest, followed by tuatara reared north of the release site (70%) and locally-reared and released tuatara survival rates were lowest (67%, result was not statistically compared). See original paper for comparisons of growth rates, post-release dispersal and home range sizes between wild-caught, locally-reared and north-reared tuatara. Juvenile tuatara originating from the same wild population were released into a predator-free fenced reserve in October–December 2012: wild-caught from an island 570 km north of the release site (14 individuals), captive-reared locally to the release site (13 individuals), and captive-reared 480 km north of the release site in a warmer climate (28 individuals). Captive-reared tuatara were hatched from artificially incubated eggs and head started until 4–6 years old. Artificial burrows were buried in the release area. Tuatara were monitored by radio-tracking for 5 months (10 wild-caught, 6 locally-reared, 10 north-reared individuals) and recapture surveys for up to 27 months after release.

(1) Nelson N.J., Keall S.N., Brown D. & Daugherty C.H. (2002) Establishing a new wild population of tuatara (*Sphenodon guntheri*). *Conservation Biology*, 16, 887–894.

(2) Towns D. (2005) Eradication of introduced mammals and reintroduction the tuatara *Sphenodon punctatus* to Motuhora (Whale Island), New Zealand. *Conservation Evidence*, 2, 92–93.

(3) Germano J.M. & Bishop P.J. (2009) Suitability of amphibians and reptiles for translocation. *Conservation Biology*, 23, 7–15.

(4) Jarvie S., Senior A.M., Adolph S.C., Seddon P.J. & Cree A. (2015) Captive rearing affects growth but not survival in translocated juvenile tuatara. *Journal of Zoology*, 297, 184–193.

14.4. Use holding pens or enclosures at release site prior to release of wild reptiles

- **Seven studies** evaluated the effects of using holding pens or enclosures at release sites prior to release of wild reptiles. Four studies were in the USA[1-3,6] and one study was in each of Australia[4], New Zealand[5] and the UK[7].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (5 STUDIES)

- **Reproductive success (1 study):** One replicated, controlled study in New Zealand[5] found that in a site where jewelled geckos were translocated into holding pens prior to release, more gravid females were found compared to a site where holding pens were not used.
- **Survival (4 studies):** Two of three controlled studies (including one replicated study) in the USA[1,2] and the UK[7] found that gopher tortoises translocated into holding pens with artificial burrows prior to release[1] or viviparous lizards released into an enclosure[7] had higher survival (recaptured) or assumed survival (dug burrows) than individuals released without pens or enclosures. The other study[2] found that translocating eastern box turtles into holding pens, or keeping them in pens for longer, did not affect post-release survival. One replicated study in the USA[6] found that survival of Florida sand skinks within holding pens with different combinations of habitat features (trees, shade cloth, woody debris) ranged from 49–79% over two years.
- **Condition (1 study):** One randomized, controlled study in the UK[7] found that viviparous lizards released into an enclosure had similar body condition compared to those released without an enclosure.

BEHAVIOUR (4 STUDIES)

- **Behaviour change (4 studies):** Two of three controlled studies (including two replicated studies) in the USA[2,3] and New Zealand[5] found that gopher tortoises[3] and jewelled geckos[5] translocated into holding pens prior to release dispersed away from the release site less frequently than those not held in pens. One study[3] also found that the activity area of tortoises held in pens was smaller in the year of release, but similar in the year after release, compared to those not held in pens. The other study[2] found that translocating eastern box turtles into holding pens, or keeping them in pens for longer, did not affect post-release dispersal behaviour. One controlled study in Australia[4] found mixed effects on a range of behavioural measures of translocating pygmy bluetongue lizards into holding pens with artificial burrows for one day compared to five days.

Background

Holding pens or enclosures at release sites may be used (sometimes termed 'soft release') to enable reptiles to become accustomed to new surroundings before release and may contain some natural habitat and burrows. Pens or enclosures may increase the chance that released animals will settle at the release site, potentially increasing the chance that the release will be successful.

This action discusses studies that test the effectiveness of placing wild-caught translocated individuals into holding pens prior to release. See also: *Use holding pens or enclosures at release site prior to release of captive-bred reptiles*.

A replicated, controlled study in 1980–1982 in five areas of pine forest in Mississippi, USA (1) found that translocations of gopher tortoises *Gopherus polyphemus* using release pens with artificial burrows prior to release were more successful than those that were not held initially in a release pen with burrows. Results were not statistically tested. When translocated gopher tortoises were initially held in release pens with artificial burrows, more tortoises were resighted or dug burrows during the 3–4-month monitoring period (17 of 21 recaptured or dug burrows, see paper for details) than when tortoises were released without a holding pen (directly released: 0 of 11; released in abandoned burrow: 1 of 5; released in artificial burrow: 0 of 3). Forty individually-marked adult gopher tortoises (some may have been captive releases) were translocated in spring–summer 1980–1982. Tortoises were released into artificial burrows in release pens (21 tortoises), directly released with no specific management (11 tortoises), into abandoned existing burrows (5 tortoises) or into artificial burrows (3 tortoises). Artificial burrows were 1 m deep and 45 degrees to the surface. Most tortoises (35 of 40) were released into areas with existing tortoise populations. Release pens were circular (4–7 m diameter), with translucent vinyl sheet walls (buried 10 cm into the ground) attached to wooden posts. Most tortoises were held in release pens for 2–4 weeks. Tortoises were monitored until late summer or early autumn in the release year.

A replicated, controlled study in 1988–1993 in a recovering area of wetlands and mixed shrubland, grasses and trees in New York, USA (2) found that releasing eastern box turtles *Terrapene carolina carolina* into holding pens prior to release did not affect post-release survival or dispersal. Annual survival was 71% and was not affected by spending time in a holding pen, neither was survival to two years (pen: 33%; no pen: 34%) or five years (pen: 27%; no pen: 24%). Post-release direction of dispersal (result presented as a bearing) and initial dispersal speed (days to disperse 100 m: average of 24–85 days) were also not affected by being in a holding pen prior to release. Nineteen gravid females, 11 clutches of 1–9 eggs, and 10 offspring were discovered following releases. In 1987–1990, a total of 335 turtles were collected from development sites or while crossing roads in suburban areas. Fifty-three turtles were fitted with radio trackers, and were either released immediately, or held in a pen for 15 days prior to release. The remaining 282 turtles were held in pens for 30 days before release. Originally a saltmarsh, the release site was created by dredge spoil deposition during 1928–1945. Radio tagged turtles were located daily for the first three days, then weekly until 1993. In 1993–1995, a trained dog *Canis lupus familiaris* was used to locate turtles.

A controlled study in 2001–2003 in a mixed forest site in South Carolina, USA (3) found that releasing translocated gopher tortoises *Gopherus polyphemus* into a holding pen prior to release resulted in less dispersal away from the release site and smaller activity areas compared to when no holding pen was used. More tortoises stayed at the released site when a release pen was used (9 months penning: 8 of 13, 62% stayed; 12 months penning: 11 of 12, 92% stayed) compared to when no pen was used (3 of 13, 23% stayed). In the year of release, tortoises held in pens for 12 months had smaller activity areas (2 ha) than those held for nine months (37 ha) or not held at all (94 ha), whereas in the year after release, activity areas were similar for all groups (5–40 ha). In 2001, tortoises were collected from an industrial development site. Groups of 12–13 adults and sub-adults were assigned either to a 'soft release' penning treatment (9 months or 12 months) or 'hard release' (no penning). All release areas contained 24 starter burrows. All turtles were released in 2002 and relocated in October–November 2002 and March–October 2003. Dispersers were retrieved and re-released at the release site.

A controlled study in 2009 in a grassland enclosure in South Australia, Australia (4) found that translocated pygmy bluetongue lizard *Tiliqua adelaidensis* confined to holding pens with artificial burrows for one day after release dispersed to marginal habitat less frequently and basked more than lizards confined for five days. After translocated pygmy bluetongue lizards were released from holding pens, lizards confined to a pen for one day dispersed to marginal habitat less frequently (0.2 lizards/cage/day) and basked for longer (22 minutes/hour) than lizards confined for five days (dispersal: 0.8 lizards/cage/day; basking time: 13 minutes/hour). Activity levels, movements, burrow switching, and agonistic interactions were similar between lizards confined for one or five days (see original paper for details). In October 2009, sixteen pygmy bluetongue lizards were captured in the wild and released into one of four predator-proof cages in a zoo enclosure (4 lizards/cage). Each cage included a central grassy circle (4 m diameter) with artificial burrows, surrounded by a strip of bare ground (5 m wide), encircled by a strip of marginal habitat (0.5 m wide) containing artificial burrows (see original paper for details of burrows). When lizards were released, all cages had a holding pen around the central grass areas. The pen was removed from two cages after one day and from the remaining two cages after five days. Lizard activity was monitored by video cameras over 10 days and analysis of lizard behaviour was based on observations from days 6–10 of the study.

A replicated, controlled study in 2011–2012 in the Orokonui Valley, New Zealand (5) found that keeping translocated jewelled geckos *Naultinus gemmeus* in a holding pen for up to ten months prior to release resulted in less movement away from their release site compared to unpenned geckos. None of the penned geckos (10 individuals) moved outside of the release area after the pen was removed (distance moved from release site: 1–16 m) compared to 67% (six of nine geckos) of the unpenned geckos (distance moved from release site: 4–39 m). Fourteen months after release, four females (all gravid) were found at the penned site and two (neither gravid) were found at the unpenned site. Forty-two geckos were translocated to Orokonui Ecosanctuary in December 2011 and January 2012 (21 females, six males and 15 unsexed juveniles) and held in a release pen (10–15 m wide, 55–60 m long and 0.5 m high) until September 2012, at which point the pen was removed. In September 2012, eleven individuals (six females, three males, two unsexed subadults) were released directly at a nearby site (200 m away). Ten penned and nine unpenned geckos were monitored by radio tracking (attached using a 22 x 3 cm self-adhesive fabric strip) one to two times daily for three weeks.

A replicated study in 2008–2010 in dry scrubland in Florida, USA (6) found that populations of Florida sand skinks *Plestiodon reynoldsi* translocated away from a proposed mining site and released into enclosures with different habitat features (trees, shade, woody debris) survived at least three years. Estimates of overall survival of translocated skinks ranged from 49–79%, and 105 of 300 skinks were recaptured during the two years following release. Provision of shade may have been important in explaining skink survival (reported as model result but effect size not reported). Newborn skinks (19 in 2008, 13 in 2009) were captured in all enclosure types. A further 35 newborns were trapped in 2010 (unpublished data). Skinks were sourced in spring 2007 from a site scheduled for sand mining and released in to fifteen 20 m^2 enclosures (20 lizards/enclosure). Enclosures had five experimental treatments (tree only, shade cloth only, tree and coarse woody debris added, coarse woody debris only, control with no shade or debris). Skinks were trapped in enclosures in spring 2008–2009 (16 drift fences and 76 bucket-traps/enclosure), and further trapping was carried out in 2010 (method not given).

A randomized, controlled study in 2016–2017 in an area of mixed grassland, scrub and woodland in Kent, UK (7) found that more translocated viviparous lizards *Zootoca vivipara* were recaptured after release into an enclosure compared to those released in an unenclosed area, and that body condition was similar in the enclosed and unenclosed areas. More lizards were resighted after release in the enclosure (101 lizards) than in the unenclosed area (16 lizards). Body condition was similar for lizards in the enclosure and those in the unenclosed area (reported as condition index). Two adjacent sites in the wider release area (1.5 ha each) were selected, and one was randomly selected and enclosed with a reptile-proof fence (38 cm high, buried 30 cm deep). Both sites were provisioned with one hibernaculum and four earth banks. In 2016, lizards were translocated to both sites in the release area (total of 1,364 lizards) and 695 were released in the enclosure and 669 were released in the unenclosed area. In April–May 2017, translocated lizards were monitored at the two sites using visual encounter surveys and artificial cover boards (45/site).

(1) Lohoefener R. & Lohmeier L. (1986) Experiments with gopher tortoise (*Gopherus polyphemus*) relocation in southern Mississippi. *Herpetological Review*, 17, 37–40.

(2) Cook R.P. (2004) Dispersal, home range establishment, survival, and reproduction of translocated eastern box turtles, *Terrapene c. carolina*. *Applied Herpetology*, 1. 197–228.

(3) Tuberville T., Clark E., Buhlmann K. & Gibbons J. (2005) Translocation as a conservation tool: Site fidelity and movement of repatriated gopher tortoises (*Gopherus polyphemus*). *Animal Conservation*, 8, 349–358.

(4) Ebrahimi M. & Bull C.M. (2013) Determining the success of varying short-term confinement time during simulated translocations of the endangered pygmy bluetongue lizard (*Tiliqua adelaidensis*). *Amphibia-Reptilia*, 34, 31–39.

(5) Knox C.D. & Monks J.M. (2014) Penning prior to release decreases post-translocation dispersal of jewelled geckos. *Animal Conservation*, 17, 18–26.

(6) McCoy E.D., Osman N., Hauch B., Emerick A. & Mushinsky H.R. (2014) Increasing the chance of successful translocation of a threatened lizard. *Animal Conservation*, 17, 56–64.
(7) Nash D.J. (2017) An assessment of mitigation translocations for reptiles at development sites. PhD thesis, University of Kent, University of Kent.

14.5. Release reptiles into burrows

- **One study** evaluated the effects of releasing reptiles into burrows on their populations. This study was in the USA[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Survival (1 study):** One replicated study in the USA[1] found that both releasing translocated gopher tortoises into abandoned or artificial burrows or releasing without burrows had low success, but providing burrows inside release pens resulted in more successful translocations.

BEHAVIOUR (0 STUDIES)

Background
Some reptiles rely on burrows to provide shelter from predators or weather extremes. Providing burrows as part of releasing translocated reptiles increases the suitability of the release habitat and reduces the immediate requirements on individuals (which may already be stressed from the translocation process) to find suitable shelter in a novel environment.

A replicated study in 1980–1982 in five areas of pine forest in Mississippi, USA (1) found that both releasing gopher tortoises *Gopherus polyphemus* into abandoned or artificial burrows and releasing tortoises with no burrows had low success, but providing burrows inside of release pens tended to result in more successful translocations. Results were not statistically tested. Success of translocations of tortoises placed in abandoned burrows or artificial burrows without release pens was low (Abandoned: 1 of 5 successful; artificial: 0 of 3), as was success of releases without a burrow or pen (0 of 11). When translocated gopher tortoises were initially held in release pens with artificial burrows, 17 of 21 translocations were successful. Forty individually-marked adult gopher tortoises (some may have been captive-bred) were translocated in spring–summer 1980–1982 (one tortoise = one translocation). Tortoises were released into either abandoned existing burrows (5 tortoises), artificial burrows (1 m deep; 3 tortoises), artificial burrows in circular release pens for 2–4 weeks (4–7 m diameter pens; 21 tortoises) or were directly released with no specific management (11 tortoises). Tortoises were monitored until late summer or early autumn in the release year and translocations were judged successful if previously abandoned burrows became active and a translocated tortoise was found in them, or new tortoise burrows were dug in areas without pre-existing tortoise populations.

(1) Lohoefener R. & Lohmeier L. (1986) Experiments with gopher tortoise (*Gopherus polyphemus*) relocation in southern Mississippi. *Herpetological Review*, 17, 37–40.

Mitigation translocations

14.6. Translocate problem reptiles

- **Seven studies** evaluated the effects on reptile populations of translocating problem individuals. Two studies were in each of Australia[2,7] and Canada[3,5], one was in each of the USA[1] and Hong Kong[6] and one was conducted across multiple countries[4].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (6 STUDIES)

- **Abundance (1 study):** One global review[4] found that when using recruitment to the adult population as a measure of success, mitigation translocations (of both problem reptiles and moving away from threats) failed more often than those carried out for conservation or research purposes.
- **Survival (5 studies):** Two of four controlled studies (including two replicated studies) in Australia[2,7], Canada[5] and Hong Kong[6] found that survival of translocated problem tiger snakes[2] and massasauga rattlesnakes[5] was similar to resident snakes for six months[2] or until hibernation[5]. One study[7] found that more translocated problem dugite snakes died within two months than resident snakes. The other study[6] found mixed effects on survival of translocating problem white-lipped pit vipers compared to resident snakes. One controlled study in the USA[1] found that two of seven translocated problem Gila monsters died within 1–24 months of translocation.

BEHAVIOUR (4 STUDIES)

- **Behaviour change (4 studies):** Two controlled studies (including one replicated study) in Australia[2,7] found mixed effects on movement behaviour[2,7] and home range size[2] of translocating problem tiger snakes[2] and dugite snakes[7] compared to resident snakes. One randomized, controlled study in Canada[5] found that translocated massasauga rattlesnakes moved further from release points after two days than snakes released at their point of capture, but distances were similar after 18 days. One controlled study in the USA[1] found that the home ranges of problem Gila monsters translocated >1 km were similar in size to those translocated <1 km.

OTHER (5 STUDIES)

- **Human-wildlife conflict (5 studies):** Three of five controlled studies (including three replicated studies) in the USA[1], Australia[2], Canada[3,5] and Hong Kong[6] of translocations of problem Gila monsters[1], tiger snakes[2] and western rattlesnakes[3] found that at least some returned to their point of capture[1,3] or moved in to adjacent suburban areas[2] within a month. One of the studies[1] found that while all problem Gila monsters translocated <1 km returned to their point of capture, none of those translocated >1 km returned. The other two studies[5,6] found that no massasauga rattlesnakes[5] or problem white-lipped pit vipers[6] returned to their point of capture.

Background

Some species may come into conflict with humans due to real or perceived risks of interacting with them (e.g. venomous or carnivorous reptiles) or due to nuisance behaviours. Retaliation by humans may limit the persistence of these species close to human settlements. Relocating individuals to alternative locations where the potential for interaction with humans is reduced may be desirable in situations where the alternative is euthanasia. Typically, mitigation translocations of problem individuals involve moving low numbers of animals a minimum necessary distance away from where they were collected.

A controlled study in 2000–2002 in an urban desert setting in Arizona, USA (1) found that no translocated problem Gila monsters *Heloderma suspectum* moved >1 km returned to their point of capture, but individuals translocated <1 km all returned. Zero of seven lizards translocated 2–25 km returned to their point of capture, whereas all 18 lizards translocated 0–1 km returned within 1–30 days. Two of 25 died during the 1–24-month monitoring period (1: translocated 2 km, survived 15 months; 2: translocated <1 km, survived 19 months) and five were lost (translocated 7–22 km). Home ranges of long and short distance translocated individuals were statistically similar (long-distance: 8–190 ha; short-distance: 2–37 ha). In 2000–2001, problem Gila monsters were obtained following calls from residents. Lizards were surgically implanted with radio-transmitters and translocated 0–25 km from their point of capture (< 1 km: 18 lizards; > 1 km: 7 lizards). Lizards were located every 2–3 days in March–October and 3–5 days in November–February in 2000–2002.

A replicated, controlled study in 2002–2003 in a grassy, wooded parkland close to suburban areas in Victoria, Australia (2) found that translocated problem tiger snakes *Notechis scutatus* had similar survival compared to resident snakes but moved longer distances and often returned to surrounding suburban areas. Survival rates were similar for translocated (7 of 8, 88%) and resident snakes (4 of 6, 67%) over six months. Movement between re-sightings of translocated snakes was larger than residents (translocated: 140 m; resident: 64 m) and half of translocated snakes moved out of the release site into adjacent suburban areas within 1–16 days. Translocated snakes had larger home ranges than residents (translocated: 28 ha; resident: 5 ha), but their core ranges (translocated: 1 ha; resident: 1 ha) and total area visited (translocated: 22 ha; resident: 4 ha) were statistically similar. Eight translocated snakes (four females, four males; trapped within 5 km of release site) and six resident snakes (two female, four males; released at point of capture) were released within the 123 ha parkland area. They were surgically implanted with radio transmitters and tracked 2–5 times/week between spring (October) 2002 and autumn (March) 2003.

A replicated, controlled study in 2004–2005 in one valley of shrubs and coniferous woodland in British Columbia, Canada (3) found that short distance translocations of problem western rattlesnakes *Crotalus oreganus* did not prevent most snakes returning to areas of human activity. Twelve of 14 (86%) translocated snakes returned to the area in which they were captured within an average of 20 days. Snakes returned to areas of human activity 1–7 times each (average of 3). Two snakes from the translocated group were killed by humans (2 of 14, 14%), whereas no mortality was observed in the group of snakes away from humans. In April 2004–August 2005, fourteen adult male rattlesnakes were monitored (by radiotracking) in an area with human activity (235 ha) and 14 were monitored in an area without human activity (235 ha). When a snake was found in an area of human activity, it was translocated a short distance (average 500 m) to a habitat free of human development. In 2004–2005, snakes were located every two days during the active season (April–October), with five individuals tracked during both years.

A review of worldwide reptile translocation projects during 1991–2006 (4) found that translocations carried out because of human-wildlife conflict (mitigation translocations) failed more often than those carried out for conservation or research purposes. Translocations to mitigate impacts of "problem" reptiles and building and development were combined. Projects that translocated animals due to human-wildlife conflicts failed more often (63% of 8 projects) than those for conservation purposes (15% of 38) and those for research purposes (50% of 5). Success was independent of the life-stage translocated, number of animals released and geographic region. Mitigation translocations included those used to deal with "problem" animals, as well as building and development mitigation. Success was defined as evidence of substantial recruitment to the adult population during monitoring over a period at least as long as it takes the species to reach maturity.

A randomized, controlled study in 2003 in a temperate forested site in Ontario, Canada (5) found that short-distance translocations did not affect massasauga rattlesnake *Sistrurus catenatus catenatus* survival. A similar number of translocated snakes and snakes released at point of capture survived until hibernation (translocated: 4 of 5, 80%; point of capture releases: 8 of 9, 89%). Translocated snakes moved further from their release site after two days (150 m) than snakes released at point of capture (50 m), but distance from release site was similar after 18 days (translocated: 330 m; point of capture: 270 m). No translocated snakes returned to their capture location. Rattlesnakes were captured in July 2003 and translocated either 200 m in a random direction (one female, four males) or released at point of capture (three females, six males). Such short distance translocations are commonly carried out for problem snakes. All snakes were implanted with radio transmitters and relocated every two days for 18 days.

A replicated, controlled study in 2012–2013 in sites of mixed shrubland and mixed forest in the Hong Kong, China (6) found that translocating problem white-lipped pit vipers *Cryptelytrops albolabris* away from human settlements resulted in lower survival compared to resident snakes in one of two years, but no translocated snakes returned to their point of capture. In 2012, a similar number of snakes died from the translocated (6 of 8, 75%) and resident (5 of 7, 71%) groups (result was not statistically tested). In 2013, more translocated snakes died than did residents (translocated: 9 of 12, 75%; resident: 3 of 11, 27%). No translocated snakes showed homing behaviour towards their point of capture. Forty-one problem snakes were captured near human settlements and released 3–30 km away. In 2012, translocated snakes were released in a site of mixed shrub and grassland, and in 2013, they were release in a woodland site. Resident snakes were all captured in woodland sites. Vipers were located 1–3 times/week for 18 (translocated) and 31 weeks (resident) in 2012, and 26 weeks (all snakes) in 2013.

A controlled study in 2015–2017 in a suburban area in Perth, Australia (7) found that translocated problem dugite snakes *Pseudonaja affinis* (urban or problem individuals) had larger activity ranges and higher mortality rates than resident snakes. Translocated snakes had larger maximum activity ranges (11 m²/day) compared to resident snakes (1 m²/day). Translocated snakes travelled similar distances (31 m/day) to resident snakes 11 m/day). All translocated snakes died during the study (4 of 4 individuals) compared to half of the resident snakes (3 of 6 individuals). Deaths were caused by predation or road collisions. In total 10 snakes (six resident snakes and four translocated snakes) were tracked for up to 2 months each in September–December 2015–2017. Snakes were either caught opportunistically in urban areas (two individuals) or reported as problem individuals (eight individuals).

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(3) Brown J.R., Bishop C.A. & Brooks R.J. (2009) Effectiveness of short-distance translocation and its effects on western rattlesnakes. *The Journal of Wildlife Management*, 73, 419–425.
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14.7. Translocate reptiles away from threats

Background

Translocations are sometimes carried out to remove individuals from specific threats within their range, for example away from development areas ('mitigation translocation'). Mitigation translocations may be carried out as a preventative measure to protect individuals but have been criticized for prioritising the process of removing individuals above establishing viable populations of translocated individuals in the destination location (Sullivan *et al*. 2014). A number of issues should be carefully considered before carrying out such translocations, including whether the proposed release site contains suitable habitat; whether the release of additional animals at an occupied site could negatively impact on the resident population; and whether a translocation alone can mitigate the impact of losing suitable habitat due to a development or other threat.

Due to the number of studies found, this action has been split by species group, though no studies were found for amphisbaenians.

For studies where individuals are relocated for short periods to mitigate risks posed by temporary threats (e.g. habitat management) see *Temporarily move reptiles away from short-term threats.*

Sullivan B.K., Nowak E.M. & Kwiatkowski M.A. (2014) Problems with mitigation translocation of herpetofauna. *Conservation Biology*, 29, 12–18.

Sea turtles

- We found no studies that evaluated the effects of translocating sea turtles away from threats on their populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Tortoises, terrapins, side-necked & softshell turtles

- **Nine studies** evaluated the effects of translocating tortoises, terrapins, side-necked & softshell turtles away from threats on their populations. Seven studies were in the USA[1-4,6-8], one was in France[9] and one was global[5].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (9 STUDIES)

- **Abundance (1 study):** One global review[5] found that when using recruitment to the adult population as a measure of success, mitigation translocations (both away from threats and moving problem reptiles) failed more often than those carried out for conservation or research purposes.
- **Reproductive success (2 studies):** One replicated, controlled study in the USA[7] found that desert tortoises translocated away from development areas produced a similar number of eggs compared to resident tortoises over 2–3 years. One replicated study in the USA[1] found that eastern box turtles translocated away from developments and suburban areas reproduced successfully at the release site.
- **Survival (8 studies):** Three of four studies (including three controlled studies) in the USA[4,7,8] and France[9] found that survival of desert tortoises[7,8] or Hermann tortoises[9] translocated away from developments was similar compared to resident tortoises for 2–3 years following release. The other study[4] found that survival in the year of release of 74 gopher tortoises translocated away from a development was lower than for established tortoises from a previous translocation. Three studies (including one replicated, controlled study) in the USA[1,2,3] found that eastern box turtles[1], gopher tortoises[2] and desert tortoises[3] translocated away from developments survived for varying durations over monitoring periods of one[2] to five years[1]. One study in the USA[6] found that at least 20% of 106 gopher tortoises translocated away from a development site survived the over-wintering period and at least two did not.
- **Condition (1 study):** One controlled study in the USA[8] found that desert tortoises translocated away from an energy plant had higher body temperatures compared to resident tortoises in the first year after release, but similar temperatures in the next two years.

BEHAVIOUR (2 STUDIES)

- **Use (1 study):** One replicated study in the USA[1] found that 47% of eastern box turtles translocated away from developments or suburban areas established home ranges at the release site whereas 25% left the site. One controlled, before-and-after study in

France[9] found that Hermann tortoises rescued from a development and translocated in autumn took longer to establish home ranges than those translocated in spring.

- **Behaviour change (1 study):** One replicated, controlled study in the USA[7] found that desert tortoises translocated away from developments moved more than resident tortoises.

A replicated study in 1988–1995 in a recovering area of wetland and mixed shrubland, grasses and trees in New York, USA (1) found that translocating eastern box turtles *Terrapene carolina carolina* away from developments and suburban areas resulted in some turtles surviving at least five years and reproducing. Annual survival was estimated at 71%, and of the 53 radio–tracked translocated individuals, 13 (25%) left the site, 25 established home ranges (47%; 17 in the release year, two in year 1, three in year 2, three in year 3) and 15 died (28%). Nineteen gravid females, 11 clutches of 1–9 eggs, and 10 offspring were also found following releases. In 1987–1990, a total of 335 turtles were collected, either from development sites or while crossing roads in suburban areas. Fifty-three turtles were fitted with radio trackers, and were either released immediately, or held in a pen for 15 days prior to release. The remaining 282 turtles were held in pens for 30 days before release. Originally a saltmarsh, the release site was created by dredge spoil deposition during 1928–1945. Radio tagged turtles were located daily for the first three days, then weekly until 1993 (when radio tracking ceased). In 1993–1995, a trained dog *Canis lupus familiaris* was used to locate turtles.

A study in 2001–2003 in a mixed forest site in South Carolina, USA (2) found that most gopher tortoises *Gopherus polyphemus* translocated away from a development site survived at least one year after release. Thirty-four of 38 (86%) adult or sub-adult tortoises survived at least one year and the remaining four were lost within 15 days of release. More tortoises stayed at the released site when a release pen was used (9 months penning: 8 of 13, 62% stayed; 12 months penning: 11 of 12, 92% stayed) compared to when no pen was used (3 of 13, 23% stayed). In 2001, tortoises were collected from an industrial development site. Groups of 12–13 adults and sub-adults were assigned either to a soft release penning treatment (9 months or 12 months) or hard release (no penning). All release areas contained starter burrows. All turtles were released in 2002 and relocated in October–November 2002 and March–October 2003. Dispersers were retrieved and re-released at the release site.

A replicated, controlled study in 1997–1998 in a site of desert scrub in southern Nevada, USA (3) found that most desert tortoises *Gopherus agassizii* translocated away from development areas and held in pens for 2–7 years prior to release survived at least two years after their release. Overall, six of 28 (21%) tortoises died in the first year following release, and no tortoises died in the second year. Mortality rates were similar between tortoises receiving supplementary water (4 of 15, 27%) and those not supplemented (2 of 13, 15%). Released tortoises were held in outdoor pens for two years (juveniles) or seven years (adults) after being removed from areas undergoing urban development. One to two months prior to release, tortoises either received supplementary water (sprinklers on for 15 minutes/day and saucers placed to catch water) (6 females, 8 males, 1 juvenile) or received no water (7 females, 5 males, 1 juvenile). Tortoises were released into artificial burrows in April–May 1997, and the release site was fenced off from a nearby road. Tortoises were relocated by radio-tracking through July 1997 to November 1998.

A study in 1994–2008 on a grassy island in Georgia, USA (4) found that gopher tortoises *Gopherus polyphemus* translocated away from a development site and provided with a starter burrow had lower initial survival than established tortoises from a previous translocation. Twenty-eight of 76 (37%) newly released tortoises were never recaptured. Initial survival was estimated to be lower for newly released adults (1st year: 75%) compared to for established adults (98%), and lower for newly released immature tortoises (1st year: 45%, 2nd year: 79%) compared to established immature tortoises (84%). In 1994, seventy-four tortoises (23 males, 32 females, 19 unsexed immature tortoises) were translocated from a development site in Georgia, USA. Each was permanently marked with unique notches on the shell and PIT tags and provided with a starter burrow. Between 1987–1993, a total of 25–30 unmarked tortoises of unknown origin were released on the island and not marked until 1994. Turtles were trapped twice a year by bucket or wire traps in autumn and spring from 1994–1998.

A review of worldwide reptile translocation projects during 1991–2006 (5) found that translocations of reptiles away from threats and translocations of ‘problem’ reptiles (mitigation translocations) failed more often than those carried out for conservation or research purposes. Translocations to mitigate impacts of building and development and ‘problem’ reptiles were combined. Mitigation translocations failed more often (63% of 8 projects) than those for conservation purposes (15% of 38) and those for research purposes (50% of 5). Success was independent of the life-stage translocated, number of animals released and geographic region. Mitigation translocations included building and development mitigation as well as those used to deal with “problem” animals. Success was defined as evidence of substantial recruitment to the adult population during monitoring over a period at least as long as it takes the species to reach maturity.

A study in 2001–2005 in a site of mixed forest in South Carolina, USA (6) found that translocating gopher tortoises *Gopherus polyphemus* away from a development site to the northerly part of their range resulted in most tortoises surviving the overwinter period. Of 21 tortoises fitted with temperature loggers, all survived the overwintering period. Two tortoises (both immature individuals) that were not fitted with loggers died during the overwintering period. In August–October 2001, a total of 106 tortoises (39 adults, 32 juveniles, 35 hatchlings) were collected from an industrial development site and translocated to an area where they were historically abundant but were absent at the time of translocation. A prescribed burn took place every three years in the release area, with the most recent in spring 2001. Tortoises were released in October 2001 or spring 2002. Twenty-one tortoises were fitted with temperature loggers, which monitored tortoise temperatures during the winters of 2002–2003 and 2004–2005.

A replicated, controlled study in 1997–2000 in five sites of desert scrub in Utah, USA (7) found that Agassiz’s desert tortoises *Gopherus agassizii* translocated away from development areas had similar survival and produced a similar number of eggs as resident tortoises. Overall annual survival was high (94%), and 89% of translocated tortoises (141 of 159) and 85% of resident tortoises (61 of 72) survived for at least 2–3 years. Time spent in captivity (15–2,300 days) did not affect survival (see paper for details). Translocated and resident tortoises produced a similar number of eggs during the study (2–8 eggs/female). Overall, translocated tortoises moved more than residents (translocated: 1,600 m; resident: 600 m). Translocated tortoises were sourced from two facilities used to house tortoises displaced by urban development (held for 15–2,300 days) and 120 individuals were translocated to a total of five sites over three years (17–82 tortoises/year to 1–4 sites in 1997–1998). Translocated tortoises were compared to 72 resident tortoises randomly encountered at two sites in Nevada. All tortoises were marked and monitored weekly using radio telemetry.

A controlled study in 2010–2014 in desert scrubland in southern California, USA (8) found that Mojave desert tortoises *Gopherus agassizii* translocated away from an energy plant and held in captivity for two years had similar annual mortality risk, but higher body temperatures in the first year after release, compared to wild resident tortoises. Mortality risk was similar between translocated tortoises (5% mortality/year) and wild resident tortoises (3–5% mortality/year). Translocated tortoises had higher average maximum daily body temperatures (36.8°C) compared to wild resident tortoises (35.9°C) in the first year after translocation and spent more time above 35°C (113 minutes) than resident tortoises (76–84 minutes). Translocated tortoise temperatures were similar to wild resident tortoises in the second and third year after translocation (see paper for details). In October 2010, tortoises were collected from near a thermal energy plant and maintained in captivity until April 2012, when they were released into an 8,798 ha area adjacent to the energy plant (<500 m from the centre of their previous home range). Translocated tortoise survival and body temperatures were compared to resident tortoises in the release area and resident tortoises from two nearby areas with similar habitat (resident tortoises caught for monitoring in spring–autumn 2011). All tortoises (351 total individuals) were radio-tracked in April–September 2012–2014 and a subset (55 translocated, 73 residents in release area, 87 nearby residents) were fitted with temperature loggers which were monitored between April 2012–September 2014.

A controlled, before-and-after study in 2012–2016 in mixed scrub and woodland in south-eastern France (9) found that Hermann tortoises *Testudo hermanni hermanni* rescued from developments that were rehabilitated and translocated had similar survival over two years compared to wild tortoises, and tortoises released in spring established home ranges more quickly than tortoises released in autumn. Over two years after release, average survival of rehabilitated, translocated tortoises (83–86%, 24 individuals) was similar to wild tortoises (93–100%, 31 individuals). Autumn-released rehabilitated, translocated tortoises took longer to establish home ranges (258 days) than those released in spring (139 days). Rehabilitated, translocated tortoises settled similar distances from release locations regardless of season of release (see original paper for details). In total, 24 rehabilitated (with various injuries or rescued from urban developments) Herman tortoises were translocated in April 2013 (12 individuals) and October 2013 (12 individuals) and radio tracked. Twenty resident tortoises and 11 from another population were also radio tracked in the release area, and six were tracked from a separate population in 2012–2015.

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Snakes and lizards

- **Nine studies** evaluated the effects of translocating snakes and lizards away from threats on their populations. Four studies were in the UK[1,2,5,9], two were in New Zealand[7,8], one was in each of South Africa[3] and the USA[6] and one was global[4].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (9 STUDIES)

- **Abundance (3 studies):** One review of lizard mitigation translocation projects in New Zealand[8] found that one of 28 projects found evidence of population growth following release. One global review[4] found that when using recruitment to the adult population as a measure of success, mitigation translocations (both away from threats and of problem reptiles) failed more often than those carried out for conservation or research purposes. One replicated study in South Africa[3] found that 2–5 years after translocating black-headed dwarf chameleons to two sites away from a development site, one site hosted more chameleons than were released, whereas the other hosted less.
- **Reproductive success (4 studies):** One review of lizard mitigation translocation projects in New Zealand[8] found that one of eight projects found evidence of breeding following release. One controlled study in the UK[1] and one replicated study in New Zealand[7] found that following translocation away from a development site[1] or from the threat of poaching[7], 14–15% of female slow worms[1] and jewelled geckos[7] were found to be gravid within 12–14 months following release. One study in the UK[5] found that

following a translocation of 119 adders away from flood defence works, one neonate was observed within six months of release.

- **Survival (6 studies):** Five studies (including two replicated studies) in the UK[1,2,5], the USA[6] and New Zealand[7] found that slow worms[1], common lizards[2], adders[5], skinks released in to enclosures[6] and jewelled geckos[7] translocated away from threats survived for varying durations over monitoring periods that lasted from six months[2] to two years[1]. One site comparison study in the UK[9] found that 20 years after slow worms were translocated away from a development site, annual survival was 56% for females and 23% for males.
- **Condition (2 studies):** One of two studies (including one controlled and one site comparison study) in the UK[1,9] found that slow worms translocated away from a development site had lower body mass compared to wild individuals. The other study[9] found that 20 years after slow worms were translocated away from a development site, males had higher body condition compared to wild individuals, but juveniles had lower body condition.

BEHAVIOUR (0 STUDIES)

A controlled study in 1995–1997 in site of mixed vegetation in south-east England, UK (1, same experimental set-up as 9) found that some slow worms *Anguis fragilis* translocated away from a development site survived at least two years and bred but had lower body condition compared to wild lizards. At the final release site, 62 of 103 (60%) slow worms were recaptured at least once during the first two years following release (12 males, 25 females, 25 juveniles). Five and zero gravid females were observed in 1996 and 1997 respectively, and two juveniles were presumed to be born at the release site. Translocated lizards had lower body mass for a given length than wild lizards (reported as condition index). Although 136 slow worms were originally captured in a development area and placed in a temporary enclosure, only 103 were recaptured and moved to the final release site. Slow worms (136 individuals) were relocated in 1994 from a housing development site to a 1,000 m^2 holding enclosure of grass and scrub with added hibernacula (rubble and log piles). Slow worms were recaptured under sheets of corrugated iron and translocated from July–October 1995 to a 1.7 ha island in a river that was recently cleared of overgrown vegetation; seeded with grass and native trees; and provisioned with log- and vegetation-piles and a new pond. Translocated lizards were monitored from March–October in 1996–1997 (280 visits) using corrugated iron sheets and photographs for identification and compared to a natural population 1.5 km from the island population.

A study in 2004–2005 in scrub and grassland in Suffolk, UK (2) found that after common lizards *Lacerta vivipara* were translocated away from a development site to newly constructed artificial hibernacula, lizards were still present six months later. Results were not statistically tested. Six months after lizards were first translocated to the hibernacula, both adult and juvenile lizards were observed basking around each hibernaculum. Three hibernacula (east-west ditches 20 m long, 1 m deep and 1.5 m wide with approximately 70° sloping edges) were constructed and filled with a mixture of drainage pipes, bricks, gravel, rubble, vegetation cuttings, logs and soil in autumn 2004. Plastic piping was added to facilitate lizards entering and entrances restricted in size to limit access by predators such as weasels *Mustela nivalis* and brown rats *Rattus norvegicus* (see original paper for details). The hibernacula were 60–120 m away from the development site. Approximately 70 lizards were caught and translocated in autumn 2004 and spring 2005. Lizard use of the hibernacula was monitored from March 2005.

A replicated study in 2002–2007 in two sub-tropical urban sites with mixed vegetation in KwaZulu-Natal, South Africa (3) found that after translocating black-headed dwarf chameleons *Bradypodion melanocephalum* away from a proposed development, one of two release sites hosted populations larger than the release cohort after five years. During the first year following release, fewer chameleons were found than were released at both sites (site one: 35 released, 3–22 observed; site two: 15 released, 3–12 observed). Two to five years following release, 0–5 chameleons were observed at site one, whereas 10–59 were observed at site two. Chameleons had been observed in both sites prior to the translocation, but a survey of site two in 2002 found no chameleons. In 2002, sixty-eight chameleons were captured in a proposed development area, and 35 were released at site one and 15 at site two. A barrier fence was installed between the development area and release site one. Vegetation was managed in 2004 (both sites) and 2007 (one site, see original paper for details). In 2002–2003, surveys of the specific release locations within each site were carried out at night using a torch (site one: 10 survey nights; site two: 7 survey nights). In 2004–2007, one transect was searched in site one (7 survey nights) and three were searched in site two (7 nights/transect).

A review of worldwide reptile translocation projects during 1991–2006 (4) found that translocations of reptiles away from threats and translocation of ‘problem’ reptiles (mitigation translocations) failed more often than those carried out for conservation or research purposes. Translocations to mitigate impacts of building and development and ‘problem’ reptiles were combined. Mitigation translocations failed more often (63% of 8 projects) than those for conservation purposes (15% of 38) and those for research purposes (50% of 5). Success was independent of the life-stage translocated, number of animals released and geographic region. Mitigation translocations included building and development mitigation as well as those used to deal with ‘problem’ animals. Success was defined as evidence of substantial recruitment to the adult population during monitoring over a period at least as long as it takes the species to reach maturity.

A study in 2009–2011 in grazing marshes in Norfolk, UK (5) found that some adders *Vipera berus* translocated away from flood defence works to man-made hibernacula bred, returned to the hibernacula to overwinter, and survived for at least eighteen months. Six months after translocation, up to 22 adders/day were recorded on the man-made hibernacula, including one newborn snake. Eighteen months after translocation, 21 of 119 (18%) translocated adders were sighted on or near the hibernacula. In addition, 19 new adders were observed in the vicinity. Viviparous lizards *Lacerta vivipara* (including juveniles) and grass snakes *Natrix helvetica* were also recorded on and near the hibernacula 12–18 months after they were built. In September 2009, three hibernacula (100 m approximate length; 1.5 m high, 3 m wide with 45° front and rear slopes) were constructed from natural materials on grazing marshes separated by drainage ditches (see original paper for design details). Each hibernaculum and some of the adjacent grazing area (1 ha total) was enclosed by semi-permanent fencing (plastic sheeting and wooden posts). In March 2010, a total of 119 adders were translocated from nearby flood banks that were subject to flood defence works (which took place May–October 2010). The fencing was opened from mid-May 2010. Adders were monitored in September–October 2010, March–May and July–September 2011.

A replicated study in 2008–2010 in dry scrubland in Florida, USA (6) found that populations of Florida sand skinks *Plestiodon reynoldsi* translocated away from a proposed mining site and released into enclosures survived at least three years. Estimates of overall survival of translocated skinks ranged from 49–79%, and 105 of 300 (35%) skinks were recaptured during the two years following release into the enclosures. Provision of shade may have been important in explaining skink survival (reported as model result). Newborn skinks (19 in 2008, 13 in 2009) were captured in all enclosures. A further 35 newborns were trapped in 2010 (unpublished data). Skinks were sourced in spring 2007 from a site scheduled for sand mining and released into fifteen 20 m^2 enclosures (20 lizards/enclosure). Enclosures had five experimental treatments (tree only, shade cloth only, tree and coarse woody debris added, coarse woody debris only, control with no shade or debris). Skinks were trapped in enclosures in spring 2008–2009 (16 drift fences and 76 bucket-traps/enclosure), and further trapping was carried out in 2010 (method not given).

A replicated study in 2011–2012 in the Orokonui Valley, New Zealand (7) found that some jewelled geckos *Naultinus gemmeus* translocated away from the threat of illegal collection survived for 14–24 months following release. At least 10 geckos survived for 10 months in a large holding pen following translocation. Fourteen-months after release from the holding pen or release directly into the wild, four penned females (all gravid) were found, and two direct release females (neither gravid) were found. Forty-two geckos were translocated to Orokonui Ecosanctuary from Otago Peninsula (where they were at risk of illegal collection) in December 2011 and January 2012 (21 females, six males and 15 unsexed juveniles) and held in a release pen (10–15 m wide, 55–60 m long and 0.5 m high) until September 2012, at which point the pen was removed. In September 2012, eleven individuals (six females, three males, two unsexed subadults) were released directly at a nearby site (200 m away). Ten penned and nine directly released geckos were monitored by radio tracking (attached using a 22 x 3 cm self-adhesive fabric strip) 1–2 times daily for three weeks.

A review published in 2016 of lizard mitigation translocation projects in New Zealand during 1988–2013 (8) found that most projects found evidence of breeding following release, but only one found evidence of population growth. Nine of 28 (32%) mitigation translocations had some post-release monitoring. One found evidence of population growth (more lizards found than released), eight found populations were smaller than the number released, and none resulted in complete failure (no lizards found). Only one mitigation translocated was monitored for >5 years, and breeding was observed in this population. Published and unpublished literature were searched, and key people associated with each translocation were identified and contacted for further information. Mitigation translocations were considered those motivated by removing lizards from anthropogenic threats at the donor site, including habitat destruction and illegal collection.

A site comparison study in 2013–2015 in two areas of mixed woodland and grassland in Kent, UK (9, same experimental set-up as 1) found that a translocated population of slow worms *Anguis fragilis* was still present 20 years later, and that males at the release site had higher body condition compared to males from another population, but immature slow worms had lower condition. Twenty years after release, a total of 59 slow worms were observed at the release location. Annual population estimates were 74 individuals in 2013, 44 in 2014 and 20 in 2015, and annual survival was estimated at 56% for females and 23% for males. Males at the release site had higher body condition than males from another natural population, whereas immature slow worms at the release site had lower condition than those from the natural population (results reported as condition index). In 1994, a population of 134 slow worms was translocated away from a residential development on a brownfield site and held in a temporary holding enclosure. After one year, 103 slow worms were captures from the enclosure and translocated to small island (1.7 ha) within a river. In 2013–2015, the population was monitored in April–September using artificial cover boards (53 boards: 0.5 m^2 each). Monitoring was also carried out at another location with a natural population of slow worms. Size and weight of all slow worms was measured at the time of capture.

(1) Platenberg R.J. & Griffiths R.A. (1999) Translocation of slow-worms (*Anguis fragilis*) as a mitigation strategy: a case study from south-east England. *Biological Conservation*, 90, 125–132.
(2) Showler D.A., Aldus N. & Parmenter J. (2005) Creating hibernacula for common lizards *Lacerta vivipara*, The Ham, Lowestoft, Suffolk, England. *Conservation Evidence*, 2, 96–98.
(3) Armstrong A.J. (2008) Translocation of black-headed dwarf chameleons *Bradypodion melanocephalum* in Durban, KwaZulu-Natal, South Africa. *African Journal of Herpetology*, 57, 29–41.
(4) Germano J.M. & Bishop P.J. (2009) Suitability of amphibians and reptiles for translocation. *Conservation Biology*, 23, 7–15.
(5) Whiting C. & Booth H. (2012) Adder *Vipera berus* hibernacula construction as part of a mitigation scheme, Norfolk, England. *Conservation Evidence*, 9, 9–16.
(6) McCoy E.D., Osman N., Hauch B., Emerick A. & Mushinsky H.R. (2014) Increasing the chance of successful translocation of a threatened lizard. *Animal Conservation*, 17, 56–64.
(7) Knox C.D. & Monks J.M. (2014) Penning prior to release decreases post-translocation dispersal of jewelled geckos. *Animal Conservation*, 17, 18–26.
(8) Romijn R.L. & Hartley S. (2016) Trends in lizard translocations in New Zealand between 1988 and 2013. *New Zealand Journal of Zoology*, 43, 191–210.
(9) Nash D.J. (2017) An assessment of mitigation translocations for reptiles at development sites. PhD thesis, University of Kent, University of Kent.

Crocodilians

- **One study** evaluated the effects of translocating crocodilians away from threats on their populations. This study was global[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Abundance (1 study):** One global review[1] found that when using recruitment to the adult population as a measure of success, mitigation translocations (both away from threats and of problem reptiles) failed more often than those carried out for conservation or research purposes.

BEHAVIOUR (0 STUDIES)

A review of worldwide reptile translocation projects during 1991–2006 (4) found that translocations of reptiles away from threats and translocations of 'problem' reptiles (mitigation translocations) failed more often than those carried out for conservation or research purposes. Translocations to mitigate impacts of building and development and 'problem' reptiles were combined. Mitigation translocations failed more often (63% of 8 projects) than those for conservation purposes (15% of 38) and those for research purposes (50% of 5). Success was independent of the life-stage translocated, number of animals released and geographic region. Mitigation translocations included building and development mitigation as well as those used to deal with 'problem' animals. Success was defined as evidence of substantial recruitment to the adult population during monitoring over a period at least as long as it takes the species to reach maturity.

(1) Germano J.M. & Bishop P.J. (2009) Suitability of amphibians and reptiles for translocation. *Conservation Biology*, 23, 7–15.

Tuatara

- **One study** evaluated the effects of translocating tuatara away from threats on their populations. This study was global[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Abundance (1 study):** One global review[1] found that when using recruitment to the adult population as a measure of success, mitigation translocations (both away from threats and of problem reptiles) failed more often than those carried out for conservation or research purposes.

BEHAVIOUR (0 STUDIES)

A review of worldwide reptile translocation projects during 1991–2006 (1) found that translocations of reptiles away from threats and translocations of 'problem' reptiles (mitigation translocations) failed more often than those carried out for conservation or research purposes. Translocations to mitigate impacts of building and development and 'problem' reptiles were combined. Mitigation translocations failed more often (63% of 8 projects) than those for conservation purposes (15% of 38) and those for research purposes (50% of 5). Success was independent of the life-stage translocated, number of animals released and geographic region. Mitigation translocations included building and development mitigation as well as those used to deal with 'problem' animals. Success was defined as evidence of substantial recruitment to the adult population during monitoring over a period at least as long as it takes the species to reach maturity.

(1) Germano J.M. & Bishop P.J. (2009) Suitability of amphibians and reptiles for translocation. *Conservation Biology*, 23, 7–15.

14.8. Temporarily move reptiles away from short-term threats

- **Two studies** evaluated the effects of temporarily moving reptiles away from short-term threats on their populations. One study was in France[1] and one was in Spain[2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Abundance (1 study):** One replicated, randomized, controlled, before-and-after study in Spain[2] found that after temporarily relocating Hermann's tortoises during vegetation management, a similar number were observed compared to before management began.
- **Survival (1 study):** One replicated study in France[1] found that at least 25% of temporarily relocated and released Hermann's tortoises survived for 4–5 years after re-release. The study[1] also found that 5% of individuals died while in temporary captivity.

BEHAVIOUR (0 STUDIES)

Background

Some threats to reptile habitat may be planned and take place over set time periods. This could include, for example, construction works or habitat management activities such as brush cutting which may injure or kill reptiles. In these circumstances it may be desirable to temporarily remove individuals and hold them in captivity while the planned works are carried out and then release the same animals back into their original environment.

For studies on the effect of permanently moving reptiles away from threats, see *Translocate reptiles away from threats*.

A replicated study in 1989–1990 and 1993–1994 of roadside verges in Toulon, France (1) found that almost a quarter of Hermann's tortoises *Testudo hermanni* temporarily relocated during highway construction were recaptured 4–5 years after release. Four–five years after the completion of a highway, 70 of 284 (25%) temporarily relocated and released Hermann's tortoises were recaptured. The first-year survival rate was estimated to be 51% and annual survival rate was estimated to be 78%. Most recaptured tortoises were discovered in the vicinity of their release location. While in temporary captivity, 16 of 300 tortoises died. In May 1989, a total of 300 tortoises were captured and held in an enclosure until the completion of a highway in October 1990. Tortoises were provided with supplementary food several times a week while in captivity. The new highway was fenced to limit tortoise access to the road and two culverts and a road underpass were constructed to facilitate tortoise movements. Visual searches for tortoises were carried out either side of the highway in April–October 1993–1994.

A replicated, randomized, controlled, before-and-after study in 2013–2014 in abandoned vineyards, pine and oak forest in Catalonia, Spain (2) found that after being temporarily removed and then returned after ground vegetation was cleared, western Hermann's tortoises *Testudo hermanni hermanni* were still in the area. Six months after release following vegetation cutting, five Hermann's tortoises had been observed in cleared plots compared to four tortoises before clearance (whether they were the same tortoises is not known) and single nests were laid in two of 50 cleared plots. Eighteen months after cutting, single nests were laid in five of 50 plots. In February 2013, fifty plots (100m^2 each) at three sites were cleared of shrub cover using a brush cutter. Fifteen tortoises were removed before cutting using trained detection dogs *Canis lupus familiaris* and put back afterwards. Plots were monitored for tortoises once a week in March–August 2013 and checked for nests in August 2013 and 2014.

(1) Guyot G. & Clobert J. (1997) Conservation measures for a population of Hermann's tortoise *Testudo hermanni* in southern France bisected by a major highway. *Biological Conservation,* 79, 251–256.

(2) Vilardell-Bartino A., Capalleras X., Budo J., Bosch R. & Pons P. (2015) Knowledge of habitat preferences applied to habitat management: the case of an endangered tortoise population. *Amphibia-Reptilia*, 36, 13–25.

14.9. Release reptiles outside of their native range

- **Seven studies** evaluated the effects of releasing reptiles outside of their native range on their populations. Three studies were in the US Virgin Islands[1,5,7] and one was in each of the USA[2], Mauritius[3], the Galápagos[4] and New Zealand[6].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (7 STUDIES)

- **Abundance (1 study):** One replicated study in the US Virgin Islands[1] found that following a translocation of St. Croix ground lizards to a new island, the population grew over a 10-year period.
- **Occupancy/range (2 studies):** One replicated, randomized study in the US Virgin Islands[5] found that following a release outside of their native range, St. Croix ground lizards were still present five years later. One randomized study in the US Virgin Islands[7] found that following a release outside of their native range, the area occupied by a population of St. Croix ground lizards increased from the 5th to 7th year following release.
- **Reproductive success (3 studies):** Three studies (including two replicated studies) in Maritius[3], the US Virgin Islands[5] and New Zealand[6] found that following releases outside of their native ranges, there was evidence of reproduction in released populations of Aldabra giant tortoises and Madagascar radiated tortoises[3], St. Croix ground lizards[5] and Otago skinks[6] after 11 months[6] and 5–7 years[3,5].
- **Survival (3 studies):** Two studies (including one replicated, before-and-after study) in the Galápagos[4] and New Zealand[6] found that following releases outside of their native ranges, 77% of sterilized Galápagos giant tortoises[4] and 63% of Otago skinks[6] survived for 11 months[6] or one year[4]. One study in the USA[2] found that annual survival of a second group of gopher tortoises translocated to an island was lower for newly released tortoises compared to established individuals from a previous release when the island had been outside of the native range.
- **Condition (1 study):** One replicated, before-and-after study in the Galápagos[4] found that sterilized Galápagos giant tortoises translocated outside of their native range as part of an ecological replacement gained weight during the first year following their release as.

BEHAVIOUR (0 STUDIES)

Background

Releasing species outside of their native range can be controversial, though may be considered when the former range has become unsuitable, or when populations are unable track changing environmental conditions. In addition, it may be appropriate to release animals outside of their original native range in order for them to act as ecological surrogates for analogous species that have become extinct.

This action includes studies involving translocations of wild reptiles and releases of captive-bred reptiles.

A replicated study in 2003 on an island containing forest and scrub in the US Virgin Islands (1) found that releasing St. Croix ground lizards *Ameiva polops* outside of their native range on to a newly created island resulted in a population that survived and grew over 10 years after release. Ten years after release, 21 individual lizards were identified on the island (9 adults, 11 juveniles and 1 not aged) and the total population size was estimated at 60. Ten lizards were translocated from Protestant Cay in 1990 and one from Green Cay in 1995 to the dredge spoil islet, Ruth Island (7.5 ha, made in 1965 from the construction of a shipping channel), where the species had not been present before. Lizards were visually surveyed five times from March to May 2003 on 20 randomly chosen 25 x 4 m plots in vegetated parts of the island.

A study in 1994–2008 on a grassy island in Georgia, USA (2) found that following translocation to a previously unoccupied island along with provision of starter burrows, adult gopher tortoises *Gopherus polyphemus* had higher survival than juveniles and translocated tortoises had lower initial survival than established tortoises from a previous translocation to the island. Initial survival was estimated to be lower for newly released adults (1st year: 75%) compared to established adults (98%), and lower for newly released immature tortoises (1st year: 45%, 2nd year: 79%) compared to established immatures (84%). Twenty-eight of 76 (37%) newly released tortoises were never recaptured. Between 1987–1993, between 25 and 30 unmarked tortoises of unknown origin were released on the previously unoccupied island and not marked until 1994. In 1994, a further 74 tortoises (23 males, 32 females, 19 unsexed immatures) were translocated from a development site in Georgia, USA. Each was permanently marked with unique notches on marginal scutes and PIT tags and provided with starter burrows. Turtles were trapped twice a year by bucket or wire traps placed in front of burrows in autumn and spring from 1994–1998.

A replicated study in 2006–2013 in grassland on Rodrigues Island, Mauritius (3) found that that captive-bred Aldabra giant tortoises *Aldabrachelys gigantea* and Madagascar radiated tortoises *Astrochelys radiata*, released outside of their native ranges to replace extinct tortoises and provided with supplementary food, bred in the wild. Seven years after captive-bred Aldabra giant and Madagascar radiated tortoises were released, 568 Aldabra and 1,114 radiated tortoises hatched in a private reserve. The authors reported that survival rates had been satisfactory overall. In 2006–2013, captive-bred Aldabra giant tortoises (>480 individuals) and Madagascar radiated tortoises (100 individuals) were introduced as ecological surrogates for extinct Rodrigues giant saddleback tortoise

Cylindrapis vosmaeri and Rodrigues domed tortoise *Cylindrapis peltastes* into a privately-managed 20 ha reserve. Native and endemic vegetation was planted and released tortoises were provided with supplementary food (seasonal fodder, fruit and vegetables) until replanted native vegetation matured. Any hatchlings discovered in the release area were also collected and brought into the nursery facility for up to 4 years before being returned to the release area.

A replicated, before-and-after study in 2010–2011 in grass and shrubland in the Galápagos Archipelago, Ecuador (4) found that most captive-bred hybrid adult Galápagos giant tortoises released as ecological replacements for an extinct species survived at least one year in the wild and gained weight. At least 30 of 39 (77%) translocated Galápagos giant tortoises survived one year after being released. In one year, tortoises had gained 11 kg each on average, or 22% of their body weight compared to before they were released (weight in 2011: 65 kg; weight in 2010: 54 kg). In total, 39 sterilized adult giant tortoises were introduced to Pinta Island (59 km^2) as ecological replacements for the extinct saddlebacked giant tortoise *Chelonoidis abingdonii* in May 2010. The tortoises had been maintained in captivity for all or most of their lives and were genetic hybrids (13 had domed shells and 26 had saddlebacked-type shells). Tortoises were monitored weekly in May–July 2010 (39 individuals) and up to three times in 2011 (30 individuals) using GPS loggers (20 individuals, 2–6 months of hourly data) or radio transmitters (16 individuals) or satellite GPS transmitters (3 individuals) and visual observation. Tortoises were weighed prior to release (39 individuals) and in June–July 2011 (27 individuals).

A replicated, randomized study in 2008–2013 in beach-forest on Buck Island, US Virgin Islands (5, same experimental set-up as 7) found that St. Croix ground lizards *Ameiva polops* released outside of their native range and held temporarily in enclosures, survived, bred and dispersed in the 5 years post release. In the first 71 days after translocation, 20 individually-identified St. Croix ground lizards, 32 unidentifiable individuals and one hatchling were observed in release enclosures. Five years later, adult (73% of observations) and juvenile lizards (24% of observations) were observed. Fifty-seven St. Croix ground lizards were translocated to Buck Island (71 ha) in April–May 2008, where they had not previously been present, apart from an unsuccessful translocation attempt in the 1960s. Lizards were marked, toe clipped, and held in enclosures (10 x 10 m) for 71 days after translocations began (7–8 lizards/enclosure, eight enclosures, enclosures removed in July 2008). Lizards were monitored in enclosures using visual surveys (26 x 10-minute surveys) and pitfall traps. Lizards were surveyed after one year (May–June 2009, captured by noosing) and five years (March–May 2013, visual surveys at 61 sites across the island). Invasive predators (rats *Rattus rattus* and mongoose *Herpestes auropunctatus*) were eradicated before translocation and vegetation restoration was ongoing.

A study in 2013–2014 in a man-made rock and shrub habitat in southern South Island, New Zealand (6) found that 63% of captive-bred Otago skinks *Oligosoma otagense* released outside of their known native range into a mammalian-predator-free fenced enclosure and provided with supplementary food survived at least 11 months and bred within 15 months. In total, 24 of 30 (80%) captive-bred Otago skinks survived at least three months and 19 of 30 (63%) skinks survived 11 months after release. The authors reported that 12 newborn skinks were observed in the enclosure 15 months after the skinks were released. Thirty captive-bred skinks were released into an oval outdoor enclosure (109 m^2 with an 85 cm high wooden fence) in an ecosanctuary in November 2013. The habitat was created to mimic natural Otago skink habitat and included rocky tors planted with native grass and shrubs. Skinks were photographed prior to release to enable individual identification. Skinks were monitored by observation during November 2013–February 2014 and September–October 2014, and by time-lapse photography of the enclosure (pictures were taken every 10 minutes between 0600–2100 h). Skinks were provided with supplementary food of 100 crickets/week but could also feed on invertebrates and small lizards present in the enclosure. The ecosanctuary was surrounded by a predator-proof fence and mammalian predators had almost entirely been eradicated.

A randomized study in 2013–2015 in mixed forest and scrubland on Buck Island, US Virgin Islands (7, same experimental set-up as 5) found that St. Croix ground lizards *Ameiva polops* released into restored island habitat outside of their native range increased their distribution in the fifth to seventh year after being released. Five years after St. Croix ground lizards were released, they occupied 41% of sites surveyed, six years after release, lizards occupied 60–66% of sites surveyed and seven years after release lizards occupied 74–87% of sites surveyed. Range expansion occurred in adjacent sites progressively further eastwards (see original paper for details). Fifty-seven lizards were introduced to Buck Island, where they had not previously been present, in 2008. Surveys were carried out in 63 sites (1,260 m^2 circular sites, at least 80 m apart) across the island five times/season over three days each in May 2013, May 2014, October 2015, May 2015 and October 2015. An additional 192 surveys were carried out in 32 sites in May 2013 and these sites were surveyed twice/day for three consecutive days. Vegetation restoration had been underway for 40 years and invasive predators removed prior to lizards being released.

(1) McNair D.B. & Mackay A. (2005) Population estimates and management of *Ameiva polops* (Cope) at Ruth Island, United States Virgin Islands. *Caribbean Journal of Science*, 41, 352–357.

(2) Tuberville T.D., Norton T.M., Todd B.D. & Spratt J.S. (2008) Long-term apparent survival of translocated gopher tortoises: a comparison of newly released and previously established animals. *Biological Conservation*, 141, 2690–2697.

(3) Griffiths O., Andre A. & Meunier A. (2013) Tortoise breeding and “re-wilding” on Rodrigues Island. *Chelonian Research Monographs*, 6, 178–182.

(4) Hunter E.A., Gibbs J.P., Cayot L.J. & Tapia, W. (2013) Equivalency of Galapágos giant tortoises used as ecological replacement species to restore ecosystem functions. *Conservation Biology*, 27, 701–709.

(5) Fitzgerald L.A., Treglia M.L., Angeli N., Hibbitts T.J., Leavitt D.J., Subalusky A.L., Lundgren I. & Hillis-Starr Z. (2015) Determinants of successful establishment and post-translocation dispersal of a new population of the critically endangered St. Croix ground lizard (*Ameiva polops*). *Restoration Ecology*, 23, 776–786.

(6) Bogisch M., Cree A. & Monks J.M. (2016) Short-term success of a translocation of Otago skinks (*Oligosoma otagense*) to Orokonui Ecosanctuary. *New Zealand Journal of Zoology*, 43, 211–220.

(7) Angeli N.F., Lundgren I.F., Pollock C.G., Hillis-Starr Z.M. & Fitzgerald L.A. (2018) Dispersal and population state of an endangered island lizard following a conservation translocation. *Ecological Applications*, 28, 336–347.

Captive breeding, rearing and releases (Ex-situ conservation)

14.10. Rehabilitate and release injured or accidentally caught individuals

Background

Reptiles that are injured, sick or found in a weak condition are sometimes taken in by wildlife rehabilitators, to be treated and released back into the wild. Animals may be injured or weakened due to direct interactions with human threats, for example entanglement in fishing gear, or due to natural threats such as extremes of weather caused by climate change (for example sea turtles may become 'cold-shocked' due to sudden severe cold weather). Often rehabilitation is carried out more for animal welfare reasons than for species conservation. However, for rare species it may be essential to preserve populations and release of such animals may provide opportunities for choosing where to augment populations. The success of such programmes can be difficult to judge without benchmark data for survival of wild-reared reptiles. It is also important to note that some of the studies summarized below have small sample sizes, and that unsuccessful attempts are less likely to have been reported.

Due to the number of studies found, this action has been split by species group, though no studies were found for amphisbaenians.

For studies evaluating the effect of releasing reptiles that were accidentally caught in fishing gear, see *Threat: Biological resource use – Establish handling and release procedures for accidentally captured or entangled ('bycatch') reptiles* and *Release accidentally caught ('bycatch') reptiles*.

Sea turtles

- **Four studies** evaluated the effects of rehabilitating and releasing injured or accidentally caught sea turtles on their populations. Two studies were in the USA[3,4] and one was in each of the Philippines[1] and the western Mediterranean[2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (4 STUDIES)

- **Survival (4 studies):** One study in the Philippines[1] and one controlled study in the western Mediterranean[2] found that of 79 rehabilitated sea turtles two were found dead and two alive within 1–5 months of release[1], and six rehabilitated loggerhead turtles survived for at least five months following release[2]. Two studies in the USA[3,4] found that around one third of stranded sea turtles[3] and 96% of sea turtles caught in fishing gear[4] could be rehabilitated and released. One study[3] also found that the chance of surviving the rehabilitation process varied with species.

BEHAVIOUR (1 STUDY)

- **Behaviour change (1 study):** One replicated, controlled study in the western Mediterranean[2] found that six rehabilitated loggerhead turtles showed similar behaviour to wild caught turtles across 46 of 54 comparisons.

A study in 2001–2011 in coastal fishing waters in the northeastern Sulu sea, the Philippines (1) reported that at least one of 79 rehabilitated sea turtles survived a minimum of four months after being released. Of 79 rehabilitated sea turtles, two were recaptured alive and two were found dead. One green turtle *Chelonia mydas* was recaptured alive in a fish corral an unspecified period after release. One hawksbill turtle *Eretmochelys imbricata* was recaptured alive in a fish corral 100 km from the release site 4–5 months later. One green turtle was found dead 1 km from the release site 4 months later. One olive ridley turtle *Lepidochelys olivacea* was found dead 32 km away from the release site 18 days later. In total, 79 sea turtles (green, olive ridley, leatherback *Dermochelys coriacea*, loggerhead *Caretta caretta* and hawksbill) were caught alive in fishing gear and released after a period of rehabilitation (see original paper for details). Most turtles were tagged prior to release. Turtle survival information was collected opportunistically when tagged turtles were recaptured.

A controlled study in 2003–2007 in the Balearic Islands, western Mediterranean Sea (2) found that six rehabilitated loggerhead turtles *Caretta caretta* survived for several months after return to the wild, and had largely similar behaviour to 12 wild turtles. Six rehabilitated turtles were tracked for an average of 156 days following release, and half were followed for longer than wild turtles. Rehabilitated turtles showed similar behaviour to wild turtles in 46 of 54 comparisons, with four of six rehabilitated turtles showing 1–3 behavioural differences each (see paper for details). Six injured turtles were brought to a rescue centre in 2004, 2006 and 2007 due to injuries sustained from boat strikes (2 turtles, 330–332 days in captivity), deeply embedded fishing hooks (2 turtles, 137–150 days in captivity), and injured flippers from net entanglement (2 turtles, 41 days in captivity). They were released between November 2004–March 2007. Twelve wild turtles were captured by a diver in 2003–2004 while basking. All turtles had a satellite tag attached and location data was received and processed by the Argos satellite system

A study in 1986–2004 along the coast in Florida, USA (3) found that of sea turtles found live-stranded and taken for rehabilitation, just over one third survived and were released back into the wild, and more time in rehabilitation improved the chances of turtles surviving to be released. In total, 626 (37%) sea turtles survived rehabilitation and were released back into the wild, 1,047 (62%) died in rehabilitation and 27 (2%) survived but were kept in captivity. More time spent in rehabilitation increased the likelihood of turtles surviving and being released (data presented as statistical model outputs). Most deaths occurred within a few weeks of rehabilitation and successful rehabilitation took from several months to >3 years. Loggerhead turtles *Caretta caretta* were most likely to survive rehabilitation, followed by kemp's ridley turtles *Lepidochelys kempii*, and green turtles *Chelonia mydas* had the lowest chance of survival (data presented as statistical model outputs). In 1986–2004, a total of 2,462 live-stranded sea turtles were taken into rehabilitation, of which 1,700 individuals had known outcomes and statistical modelling could be carried out using data from

392 individuals. Rehabilitated species included green, loggerhead, kemp's ridley, hawksbill *Eretmochelys imbricata*, leatherback *Dermochelys coriacea* and olive ridley *Lepidochelys olivacea* sea turtles. Turtles were all found live-stranded along the Florida coast.

A study in 2010–2014 in a coastal reef estuary in Mississippi, USA (4) found that most sea turtles accidentally caught in fishing gear were able to be released after rehabilitation, but a fifth of those animals were recaptured in fishing gear. In total, 96% of rescued sea turtles were rehabilitated and released (744 of 775 individuals). However, in the third and fourth years after the release programme began, 161 turtles were recaptured incidentally in a recreational fishery. Twenty-nine turtles were recaptured three times and two turtles were recaptured six times. Time between original release and recapture ranged from 12–1,121 days and 71% of recaptures occurred within the vicinity of the release location. In total, 775 rescued live sea turtles were brought to a rehabilitation facility in 2010–2014. The majority were incidentally caught in a recreational hook and line fishery (732 individuals) and the remainder were either caught in trawl or dredge equipment or suffering from boat strikes or live strandings. Rehabilitated turtles were released after medical clearance. Turtles were individually marked, which allowed recaptures to be monitored opportunistically as they occurred. Sea turtles caught were kemp's ridley *Lepidochelys kempii* (98%), loggerhead *Caretta caretta* (1%) or green sea turtles *Chelonia mydas* (1%).

(1) Bagarinao T.U. (2011) The sea turtles captured by coastal Fisheries in the northeastern Sulu sea, Philippines: Documentation, care, and release. *Herpetological Conservation and Biology,* 6, 353–363.
(2) Cardona L., Fernández G., Revelles M. & Aguilar A. (2012) Readaption to the wild of rehabilitated loggerhead sea turtles (*Caretta caretta*) assessed by satellite telemetry. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 22, 104–112.
(3) Baker L., Edwards W. & Pike D.A. (2015) Sea turtle rehabilitation success increases with body size and differs among species. *Endangered Species Research*, 29, 13–21.
(4) Coleman A.T., Pulis E.E., Pitchford J.L., Crocker K., Heaton A.J., Carron A.M., Hatchett W., Shannon D., Austin F., Dalton M., Clemons-Chevis C.L. & Solangi M. (2016) Population ecology and rehabilitation of incidentally captured kemp's ridley sea turtles (*Lepidochelys kempii*) in the Mississippi sound, USA. *Herpetological Conservation and Biology,* 11, 253–264.

Tortoises, terrapins, side-necked & softshell turtles

- **Four studies** evaluated the effects of rehabilitating and releasing injured or accidentally caught tortoises, terrapins, side-necked and softshell turtles on their populations. Two studies were in France[2,4] and one was in each of South Africa[1] and the USA[3].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (4 STUDIES)

- **Reproductive success (1 study):** One controlled study in France[2] found that some rehabilitated Hermann's tortoises were observed mating with resident tortoises following release.
- **Survival (4 studies):** One controlled, before-and-after study in France[4] found that survival of rehabilitated and released Hermann's tortoises was similar compared to wild tortoises over a two-year period. Three studies (including two replicated studies) in South

Africa[1], France[2] and the USA[3] found that Babcock's leopard tortoises[1], Herman's tortoises[2] and ornate box turtles[3] released following rehabilitation survived for varying durations during monitoring periods that ranged from three months[2] to 25 months[1] or until the end of the active season during the year of release[3].

BEHAVIOUR (2 STUDIES)

- **Behaviour change (2 studies):** One controlled study in France[2] found that 12 rehabilitated Herman's tortoises remained within 2 km of their release site over a three-month period. This study[2] also found that daily movement of rehabilitated and released tortoises was similar to residents. One controlled, before-and-after study in France[4] found that rehabilitated tortoises released in autumn took longer to establish a home range than those released in spring.

A replicated study in 2005–2007 in two savanna sites in northeast South Africa (1) reported that 22 Babcock's leopard tortoises *Stigmochelys pardalis babcocki* from a rehabilitation centre survived for between one and at least 25 months following release in to the wild. One tortoise survived for at least 25 months and two for 13 months. Eight tortoises were found dead 2–17 months following release. Seven were seen alive 1–17 months following release and then not seen again, and 11 were not re-seen at all. Tortoises for the release came from a rehabilitation centre. One had been confiscated from the traditional medicine trade, and the others were escaped pets. Twenty-two tortoises were released (11 males, 11 females) in January 2005, five (3 females, 2 males) in December 2006, and a further two females in February 2007. In total, 17 were fitted with radio trackers. Radio tracked tortoises were located monthly for 10 months after release, and then sporadically up to 25 months after release.

A controlled study in 2012–2013 in mountainous grasslands in Provence-Alpes-Côte d'Azur region, southwest France (2) found that 12 released rehabilitated Herman's tortoises *Testudo hermanni hermanni* survived at least three months in the wild and bred. After 3 months, all 12 released rehabilitated tortoises remained within 2 km of their release site and moved similar daily distances (27–38 m/day) to resident tortoises monitored at the same time (34–40 m/day). The authors report that female released tortoises were observed mating with male resident tortoises on several occasions. Twelve radio-tagged Herman's tortoises were released directly into a national nature reserve (165 ha) in April 2013. The released tortoises were wild individuals that had been rehabilitated and maintained in captivity in a rescue facility in naturally-vegetated outdoor enclosures (7 m x 7 m) for 2–8 years prior to release. Released tortoises were radio-tracked in April–July in 2013. Resident tortoises captured within 0.8 km of the release site were also monitored in April–July using radio-tags in 2012 (9 individuals) and 2013 (14 individuals). All tortoises were tracked daily and behaviours were observed from a distance.

A replicated study in 2008–2009 in three sites of grass and scrubland and an urban area in Texas, USA (3) found that some rehabilitated ornate box turtles *Terrapene ornata ornata* survived until the end of the activity season that they were released in. At the end of the active season, five of 17 adult and 12 of 22 hatchling/juvenile rehabilitated and released ornate box turtles were confirmed as still alive. One adult and five hatchling turtles were confirmed dead. The fate of

11 adult and five hatchling turtles was unknown. In 2008 and 2009, thirty-nine ornate box turtles (17 adults and 22 hatchlings/juveniles) were rehabilitated and released from a rescue centre to three natural and one urban locations. Turtles were radio-tagged prior to release and located 3–6 times/week during the active season, or until death or loss of a transmitter signal.

A controlled, before-and-after study in 2012–2016 in mixed scrub and woodland in south-eastern France (4) found that Hermann tortoises *Testudo hermanni hermanni* that were rehabilitated and translocated had similar survival over two years compared to wild tortoises, and tortoises released in spring established home ranges more quickly than tortoises released in autumn. Average survival of rehabilitated, translocated tortoises (83–86%, 24 individuals) was similar to wild tortoises (93–100%, 31 individuals) in the two years after release. Autumn-released rehabilitated, translocated tortoises took longer to establish a home range (258 days) than those released in spring (139 days). Rehabilitated, translocated tortoises settled similar distances from release locations regardless of season of release (see original paper for details). In total 24 rehabilitated (with various injuries or rescued from urban developments) Herman tortoises were translocated in April 2013 (12 individuals) and October 2013 (12 individuals) and radio tracked. Twenty resident tortoises and 11 from another population were also radio tracked in the release area, and six were tracked from a separate population in 2012–2015.

(1) Wimberger K., Armstrong A.J. & Downs C.T. (2009) Can rehabilitated leopard tortoises, *Stigmochelys pardalis*, be successfully released into the wild? *Chelonian Conservation and Biology*, 8, 173–184.

(2) Lepeigneul O., Ballouard J.M., Bonnet X., Beck E., Barbier M., Ekori A., Buisson E. & Caron S. (2014) Immediate response to translocation without acclimation from captivity to the wild in Hermann's tortoise. *European Journal of Wildlife Research*, 60, 897–907.

(3) Sosa J.A. & Perry G. (2015) Site fidelity, movement, and visibility following translocation of ornate box turtles (*Terrapene ornata ornata*) from a wildlife rehabilitation center in the high plains of Texas. *Herpetological Conservation and Biology*, 10, 255–262.

(4) Pille F., Caron S., Bonnet X., Deleuze S., Busson D., Etien T., Girard F. & Ballouard J.M. (2018) Settlement pattern of tortoises translocated into the wild: a key to evaluate population reinforcement success. *Biodiversity and Conservation*, 27, 437–457.

Snakes & lizards

- We found no studies that evaluated the effects of rehabilitating and releasing injured or accidentally caught snakes and lizards on their populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Crocodilians

- **One study** evaluated the effects of rehabilitating and releasing injured or accidentally caught crocodilians on their populations. This study was in India[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Reproductive success (1 study):** One study in India[1] found that found that breeding occurred in a rehabilitated and released population of mugger crocodiles four years after the first release.
- **Survival (1 study):** One study in India[1] found that seven of eight rehabilitated and released mugger crocodiles survived for at least 1–4 years after release.

BEHAVIOUR (0 STUDIES)

A study in 1977–1981 in a river with a series of pools in Andhra Pradesh, India (1) found that accidentally captured mugger crocodiles *Crocodylus palustris* that were raised in captivity before being released survived for at least 1-4 years following release. At least seven of eight released crocodiles survived for at least 1–4 years after release. All crocodiles were re-sighted at the release site, or within 100–3,000 m away. Authors reported that the first breeding took place four years after the first release. In 1977–1980, eight mugger crocodiles (5 females and 3 males) were released following rearing in captivity. Crocodiles were between 1.1–1.9 m in length at the time of release. Prior to the release, grazing of cattle and goats along the river bank, fishing and use of the area for swimming and bathing were banned. After release, crocodiles were monitored by both research staff and by staff who were there to protect the release site.

(1) Choudhury B.C. & Bustard R. (1982) Restocking mugger crocodile *Crocodylus palustris* (Lesson) in Andhra Pradesh: evaluation of a pilot release. *Journal of the Bombay Natural History Society*, 79, 275–289.

Tuatara

- We found no studies that evaluated the effects of rehabilitating and releasing injured or accidentally caught tuatara on their populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

14.11. Breed reptiles in captivity

Background

Captive breeding involves taking wild animals into captivity and establishing and maintaining breeding populations. It tends to be undertaken when wild populations become very small or fragmented or when they are declining rapidly. Captive populations can be maintained while threats in the wild are reduced or removed and can provide an insurance policy against catastrophe in the wild. Captive breeding also potentially provides a method of increasing reproductive output beyond what would be possible in the wild. However, captive breeding can result in problems associated with inbreeding depression, removal of natural selection and adaptation to captive conditions. The aim is usually to release captive-bred animals back to natural habitats, either to original sites once conditions are suitable, to reintroduce species to sites that were occupied in the past or to introduce species to new sites. Some captive populations may also be used for research to benefit wild populations.

Due to the number of studies found, this action has been split by species group, though no studies were found for amphisbaenians.

For studies that investigate the effectiveness of releasing captive-bred reptiles see *Release captive-bred reptiles into the wild*.

Sea turtles

- **Two studies** evaluated the effects of breeding sea turtles in captivity. One study was in the Cayman Islands, Costa Rica, Surinam and Ascension Island[1] and one was in Japan[2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Reproductive success (2 studies):** One replicated, controlled study in the Cayman Islands, Costa Rica, Surinam and Ascension Island[1] found that artificially incubated green turtle eggs that were laid in captivity had lower hatchling success than those laid in the wild and artificially incubated. One study in Japan[2] reported that hatching success of eggs produced by one female black turtle in captivity was 12%.

BEHAVIOUR (0 STUDIES)

A replicated, controlled study in 1969–1973 in a captive breeding facility and several sandy beaches in the Cayman Islands, Costa Rica, Surinam and Ascension Island (1) found that green turtles *Chelonia mydas* bred successfully in captivity, but hatching success was generally lower and numbers of infertile eggs higher compared to eggs taken from natural nests. Hatching success for artificially-incubated, captive-laid eggs was 42% (4,800 of 11,300 eggs) compared to 78% (76,000 of 97,300) for artificially-incubated wild-collected eggs and 88% (388 of 442) for undiscovered captive-laid eggs that incubated naturally in the breeding enclosure (result was not statistically tested). Overall, more captive-laid, artificially-incubated turtle eggs were infertile (5,800 of 11,300, 52%) than wild-collected eggs (17,500 of 97,300, 18%). By 1973, a captive facility with a sea-water breeding pool (61 x 27 m) and artificial sandy beach was stocked with 257 green turtles (captive-reared and wild-caught). Eggs laid in nests on the artificial beach (11,300 total eggs) and eggs laid in the wild in natural nests (17,500) on several beaches were collected and incubated in Styrofoam boxes (100 eggs/box, layered with muslin cloth and sand). Average incubation temperature was 28°C. Hatching success from all artificially-incubated eggs and eggs from four undiscovered captive-laid nests (442 total eggs) was evaluated after emergence.

A study in 2015–2017 on Okinawa Island, Japan (2) found that a pair of black turtles *Chelonia agassizii* bred successfully two years after being moved into a shared enclosure, though hatching success was low. In 2017, a female produced five clutches of eggs, with an average of 45 eggs/clutch. Average hatching success for three clutches laid on land was 12% and incubation periods were 52–57 days.

A further two clutches were laid in the water and all eggs were lost. A male and female turtle were acquired in 1999 and 2009 respectively. In 2015, they were both introduced to an outdoor tank (3.5 x 2.2 m) with an open water system. During the nesting season (May–August), the female was moved to a holding tank (17 x 11 x 2 m) that had an open water system and a sandy nesting area. Eggs were collected and moved to a hatchery, where sand temperatures ranged from 27–32°C.

(1) Simon M.H., Ulrich G.F. & Parkes A.S. (1975) The green sea turtle (*Chelonia mydas*): mating, nesting and hatching on a farm. *Journal of Zoology*, 177, 411–423.

(2) Kawazu I., Maeda K., Fukada S., Omata M., Kobuchi T. & Makabe M. (2018) Breeding success of captive black turtles in an aquarium. *Current Herpetology*, 37, 180–186.

Tortoises, terrapins, side-necked & softshell turtles

- **Twenty-eight studies** evaluated the effects of breeding tortoises, terrapins, side-necked & softshell turtles in captivity. Twelve studies were in the USA[2,3,5-8,19,20,22,23,25,26], four were in the Seychelles[13a,13b,18a,18b], two were in Madagascar[12,21], two were in an unknown location[9,10] and one was in each of the Galápagos[1], Germany[4], Austria[11], Jersey[14], Italy[15], India[16], China[17] and Myanmar[24].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (28 STUDIES)

- **Abundance (5 studies):** Four studies (including one replicated study) in Madagascar[12,21], the Seychelles[18b] and the USA[19] reported that captive breeding programmes produced 255 ploughshare tortoises[12], 40 and 140 giant tortoises[18b], 75 juvenile radiated tortoises[19] and 94 Madagascar big-headed turtle hatchlings[21]. One study[12] also reported that the captive population grew each year. One replicated study in Myanmar[24] reported that the number of Burmese star tortoise hatchlings produced in captivity increased from 168 to over 2,000 over eight years.
- **Reproductive success (24 studies):** Eighteen studies (including one replicated, controlled, before-and-after study) in the USA[2,3,6,7,8,20,22,23,25], the Galápagos[1], Germany[4], Austria[11], the Seychelles[13b], Italy[15], India[16], China[17] and an unknown location[9,10] reported that females produced 0–25 clutches of 1–26 eggs[2,3,6,8,9,11,13b,15,16,22,23,25], 65–78 eggs each/year[4] or a total of 10–170 eggs[1,7,10,17,20]. Three of these studies reported hatching success of 52–100%[1,10,20], four reported hatching success of 23–71%[4,8,9,11], three reported hatching success of 0–66%[9], 0–81%[16] or 0–100%[22] and six reported hatching success of 0–43%[2,7,13b,23,25] or 0–3 hatchings/clutch[3]. One other study from the Seychelles[13a] reported that 0–75% of eggs from one of two mud turtle species hatched successfully. One of the studies[4] also found that three of five eggs produced by a captive-bred tortoise hatched successfully. Two studies in Jersey[14] and the Seychelles[18a] reported that only 3 Malagasy Flat-tailed tortoise eggs[14] and 3–18 mud turtle eggs[18a] hatched successfully over 11–12 years. One study in Madagascar[21] reported that most Madagascar big-headed turtle eggs laid in captivity were infertile. One study in the USA[5] reported that hatching success of 2[nd] generation captive desert tortoises was 20–83%, whereas success for 3[rd] generation tortoises was 0–43%. One study in the USA[26] found that hatching success for captive Bourret's box turtle eggs was higher when incubated at 26–27°C compared to 28–29°C.

- **Survival (7 studies):** Three studies (including one replicated study) in the USA[8], Austria[11] and an unknown location[9] reported that 2–4 captive-bred tortoises or turtles survived for at least 28 weeks to two years. One replicated study in Italy[15] reported that all captive-bred spider tortoises survived to adult size. Two studies in the USA[7] and Jersey[14] reported that 25–30% of captive-bred tortoises died within 12–18 months. One study in the Seychelles[18a] reported that 70% of captive-bred mud turtles died during hatching

BEHAVIOUR (0 STUDIES)

OTHER (1 STUDY)

- **Offspring sex ratio (1 study):** One study in the USA[19] reported that a captive breeding programme of radiated tortoises produced 67 females and eight males.

A replicated study in 1965–1971 in a captive breeding facility in the Galápagos, Ecuador (1) found that Galápagos giant tortoise *Geochelone elephantopus hoodensis* bred in captivity. Over five nesting seasons, captive Galápagos giant tortoise females laid 19 nests in artificial nesting sites, two in natural nesting sites, and six clutches were laid on the surface of their enclosure (tortoises were unable to construct nests). For eggs that were collected and artificially incubated, 75% (24 of 32) were fertile and 63% (20 of 32) hatched successfully. In comparison, 80–86% (519 of 653 total eggs) of eggs from wild, undisturbed nests (of two other giant tortoise subspecies) were fertile, and hatching success was 76–82% (494 of 653 eggs) (results were not statistically tested). One male and ten female Galápagos giant tortoises were brought into a captive breeding enclosure to mate and nest from the 1967/1968 nesting season. In 1969/1970, 1970/1971 and 1971/1972 nesting seasons, artificial nest sites were provided (fine soil, minimum 3 m^2 and 35–40 cm deep). Nests were excavated the day after being laid and moved for artificial incubation. Hatching success was evaluated for six clutches in the 1970/1971 nesting season (1–7 eggs/clutch) and compared to 81 clutches of *Geochelone elephantopus porteri* (520 total eggs) and *Geochelone elephantopus ephippium* (133 total eggs) laid in 1969/1970–1970/1971.

A study in 1975 at Philadelphia Zoo, USA (2) reported that Galápagos giant tortoises *Geochelone elephantopus* produced one hatchling in captivity. A clutch of nine eggs was produced, one of which hatched successfully after an incubation period of 200 days. The hatchling had to be removed from the egg by hand. Four eggs were broken in the nest, and five were placed in the incubator, of which three were fertile. One was opened after 143 days and found to contain a live embryo, and one was opened after 211 days and contained a dead embryo. The adult male had been in captivity since 1928, and the adult female was hatched in captivity in 1940. Eggs were incubated at 26.7°C.

A study in 1978–1982 in the USA (3) reported that captive-born leopard tortoises *Stigmochelys pardalis babcocki* bred successfully in captivity and produced hatchlings in three of four years. From 1979–1982, a captive-born female produced 11 clutches of 1–9 eggs/clutch. The first five clutches produced no hatchlings, with most eggs breaking during laying. Subsequent clutches produced 1–3 hatchlings and incubation periods ranged from 135–202 days. In 1978–1981, a sibling pair of captive-hatched tortoises were housed together, and in 1981 an unrelated captive-hatched male was added to the pair. Tortoises were kept in an outdoor pen (5 x 3 m) when temperatures remained above 21°C during the day and 10°C at night and were otherwise housed in an indoor pen with a substrate of wood shavings. A heat-lamp was provided in the indoor pen and temperatures ranged from 21–40°C. In, 1979–1982, eggs were placed in a dry air incubator at 30°C.

A replicated study in 1976–1981 in an outdoor enclosure in Germany (4) reported that captive Hermann's tortoises *Testudo hermanni hermanni*, Greek tortoises *Testudo graeca ibera* and Russian tortoises *Agrionemys horsfieldii* bred successfully in captivity. In 1976–1981, ten females produced 65–78 eggs each year, with a hatching success of 23–71%. In 1981, a Hermann's tortoise that was hatched in captivity produced offspring (3 of 5 eggs hatched). Thirty tortoises were kept in an outdoor enclosure (180 m^2) for 7–24 years and fed a mixture of vegetables. Twenty individuals were sexually mature, including seven male and eight female Hermann's tortoises; one male and one female Greek tortoise; and two male and one female Russian tortoise.

A study in 1935–1986 in California, USA (5) found that 1st and 2nd generation captive desert tortoises *Gopherus agassizii* bred successfully in most years, but 3rd generation tortoises were successful in only two of 10 years. Authors reported only a subset of data. They estimated that the total number of eggs produced was 280 over 30 years by the 1st captive generation; 120 over 16 years by the 2nd generation; and 32 over 10 years by the 3rd generation. Reported hatching success was 20–83% for eggs produced by the 2nd generation, and 0–43% for those produced by the 3rd generation. All tortoises were descendants of an adult pair acquired in 1935 and were housed in outdoor enclosures. Eggs were collected from outdoor nests and placed in plastic bowls in 1 cm of washed sand. Bowls were covered with a damp cloth and temperatures were maintained at 26–27°C. When hatching began, eggs were moved to a sheet of waxed paper.

A study in 1977–1986 at Columbus Zoo, Ohio, USA (6) found that gibba turtles *Mesoclemmys gibba* reproduced successfully in captivity. In 1978–1982, one female produced seven clutches of 3–6 eggs. In 1985–1986, a further three females produced seven clutches of 2–7 eggs. Two of these females were offspring of the first pair. Incubation periods ranged from 140–248 days. The original male was acquired in 1968, and a female was acquired in 1977. Adults were housed along with a range of other turtle species in a 140 cm square display tank, with 50 cm deep water and a basking spot. Water temperature was 20–24°C and air temperatures were 24–32°C. Eggs were incubated at 26–31°C in sealed one-gallon jars in a 1:1 mixture of vermiculite and water (by weight), and jars were vented ever 4–6 weeks.

A study in 1965–1990 at The National Zoological Park and a private collection, Washington DC, USA (7) found that pancake tortoises *Malacochersus tornieri* bred in captivity, but most eggs produced in one population were infertile. Hatching success was four of 65 eggs (6%) in the first population and three of seven (43%) in the second. Of the remaining eggs, 46 of 65 (71%) and two of seven (29%) were infertile. Four individuals survived for at least a year or less, and one survived at least nine years. The National Zoological Park acquired its first tortoises in 1965–1972, and numbers fluctuated between 3–11 adults. The private collection acquired two females and a male in 1986–1988. Tortoises were housed in a range of different indoor enclosures and some had access to outdoor enclosures in good weather. Eggs were incubated using a range of methods (see paper for details), with average temperatures ranging from 27–31°C.

A replicated study in 1992–1993 in a captive breeding facility in the USA (8) found that parrot-beaked tortoises *Homopus areolatus* reproduced successfully in captivity. A total of nine egg clutches were produced and nine of 21 eggs (43%) hatched successfully. At least two of the hatchlings survived for ≥13 months. In 1992, six wild tortoises (3 males, 3 females) and seven captive tortoises were brought to the indoor captive breeding facility. Two habitat enclosures measuring 7 x 2 feet were constructed, and two males and 4–5 females were put in each enclosure.

A study in 1989–1992 in a captive setting [location unknown] (9) found that one of two female yellow-headed box turtles *Cuora aurocapitata* reproduced successfully in captivity. In 1992, two of three eggs produced by one female hatched successfully, and zero of three eggs from a second female hatched. Incubation lasted 64–66 days, and one hatchling was removed from the egg manually. The two hatchlings survived for at least 28 weeks. In 1989–1990, two pairs of turtles were acquired. Tanks contained a water basin (100 x 40 x 20 cm) and an island (40 x 20 cm), with water temperatures of 22°C, and air temperatures under a heating lamp at 27°C. One pair was housed together, and the second pair were kept separate. Males were introduced to both females for mating purposes. To induce egg laying, females were injected with calcium (at 60–80 mg/kg) subcutaneously in the rear leg, followed by 6 IU/kg of oxytocin intramuscularly one hour later. Eggs were placed in moist peat and incubated at 28°C at 95% humidity.

A study in 1993 in captive conditions [location unknown] (10) found that Reimann's snake-necked turtle *Chelodina reimanni* bred successfully in captivity. Captive female Reimann's snake-necked turtles were observed breeding in captivity. Four female turtles laid two–three clutches each (6–15 eggs/clutch) in one year. After artificial incubation, 43 of 74 eggs (58%) hatched successfully. The authors report that the substrate material used did not affect egg development. At least one captive-born hatchling survived at least two years. Four female and two male turtles were kept in captivity. Eggs were collected after laying and artificially incubated at a constant temperature of 28°C on a substrate of dry sand, moist vermiculite, moist perlite, or a moist sand-peat mixture.

A study in 1996–1999 in captive conditions in Vienna, Austria (11) found that tricarinate hill turtles *Melanochelys tricarinata* bred successfully in captivity. Three female turtles laid 12 clutches (1–3 eggs/clutch). Six of 23 eggs (23%) hatched and at least four hatchlings survived at least two years and five months. Four male and three female adult turtles were housed in captive facilities. Mating occurred at temperatures above 28°C. Females were x-rayed to check for pregnancy. After being laid, eggs were artificially incubated at air temperatures of 27–31°C, 85–95% humidity and on a sand-earth substrate.

A replicated study in 1986–1997 in an outdoor captive facility in north-western Madagascar (12) found that ploughshare tortoises *Geochelone yniphora* bred successfully in captivity and captive-born individuals survived at least 8–9 years in captivity. Over 10 years a captive breeding facility produced 255 surviving ploughshare tortoises and the captive population increased in size each year. The first successful captive hatching was one year after the programme began. The authors reported that mortality in captive-born juveniles was rare. In 1986, eight male and 10 female adult ploughshare tortoises were brought to an outdoor captive facility. Eggs were left to hatch in situ and after emerging, hatchlings were placed in 1 m^2 rearing enclosures until four years of age when they were moved to a larger 20 m^2 enclosure.

A study in 1997–2003 in a captive facility in Silhouette, Seychelles (13a; same experimental set-up as 18a) found that some black mud turtle *Pelusios subniger parietalis* eggs hatched in captivity, but that chestnut-bellied mud turtle *Pelusios castanoides intergularis* eggs did not hatch in captivity. In the 1997–1998 and 1998–1999 breeding seasons, no black mud turtle eggs hatched in captivity, although clutches were laid. In 1999–2000, one of 18 eggs hatched (two clutches laid), in 2000–2001, nine of 23 eggs hatched (three clutches laid), in 2001–2002, twelve of 25 eggs hatched (three clutches laid) and in 2002–2003, six of 8 eggs hatched (clutch numbers not reported). In 1999–2003, no chestnut-bellied mud turtle eggs hatched although clutches were laid in 2000–2001 (three eggs laid), 2001–2002 (two eggs laid) and 2002–2003 (24 eggs laid). The authors reported that incubation humidity was too high for chestnut-bellied mud turtle eggs. In 1997–1998, five captive black mud turtles (one male, four females) and five chestnut-bellied mud turtles (two males, three females) were brought to a captive facility (see original report for husbandry details). In 1999, four of five black mud turtles died in captivity and were replaced with five captive black mud turtles (three males, two females). In 2000–2001, two further female captive black mud turtles were added. No details of incubation are provided.

A study in 1999–2002 in naturally-vegetated outdoor captive enclosures in Silhouette Island, Seychelles (13b; same experimental set-up as 18b) found that one female Seychelles giant tortoise *Dipsochelys hololissa* and one female Arnold's giant tortoise *Dipsochelys arnoldi* successfully bred in captivity. From 1999–2001, all of the 160 eggs laid by three female Arnold's giant tortoises and all of the 47 eggs laid by a single female Seychelles giant tortoise in captivity were infertile. In 2002, three of at least 13 (23%) Arnold's giant tortoise eggs (laid by one female) and two of 21 (10%) Seychelles giant tortoise eggs (laid by one female) hatched successfully in captivity. All successfully hatched eggs were artificially incubated. Eggs reburied in the ground did not hatch and eggs left in situ were predated by crabs. The authors reported that the Arnold's giant tortoise offspring were thought to be Seychelles-Arnold giant tortoise hybrids. In 1997–1999, three male and three female Arnold's giant tortoises, four male and two female Seychelles giant tortoises, and one juvenile Aldabra tortoise *Dipsochelys dussumieri* were brought to a captive facility. In 1999–2002, three female Arnold's giant tortoises laid 21 clutches between them (6–16 eggs/clutch, two clutches with unknown clutch size) and one female Seychelles giant tortoise laid four clutches (14–21 eggs/clutch). In 2002, eggs were artificially incubated at 29–30°C.

A study in 1991–2002 at Jersey Zoo, Jersey (14) found that Malagasy Flat-tailed tortoises *Pyxis planicauda* had limited success breeding in captivity. Females produced 2–3 eggs/season, though only three eggs hatched successfully over 11 years. One hatchling died after 18 months. Incubation periods were >213, 262 and 306 days. Two females and four males were obtained in 1991, and a further three females were obtained in 1997. Males were housed in individual enclosures (50 x 50 cm), and females were housed together (400 x 50 cm enclosure). Temperature, humidity and rainfall (from a sprinkler system) were moderated to replicate the wet/dry season cycle (see paper for details). Eggs were incubated in a bowl with dry vermiculite, inside a box containing damp vermiculite (1:1 with water by weight). The incubation box was subjected to the same seasonal conditions as the captive tortoises, but temperatures were increased to 30–31°C near the end of incubation.

A replicated study in 1997–2000 in Italy (15) found that spider tortoises *Pyxis arachnoides* bred successfully in captivity. Females produced three clutches/year each of one egg/clutch, and 25% hatched successfully. All hatchlings survived to adult size. Tortoises were imported from Madagascar in 1997–1998 or were obtained from private breeders or other facilities. Reproduction was monitored in captivity over two years. Some data were obtained from private breeders.

A study in 2001–2009 in a captive setting in Uttar Pradesh, India (16) reported that red-crowned roof turtles *Batagur kachuga* bred successfully in captivity. Four females produced 1–5 clutches/year of 11–23 eggs, and hatching success ranged from 0–81%. In 2001, four female and two male turtles were acquired. They were quarantined for six months before being introduced to an enclosure with a large pond (30 x 15 m) with a number of other turtles of different species. In 2003–2009, the nesting mound was searched frequently, and eggs that were found were removed and incubated in plastic boxes with moist sand.

A replicated, controlled, before-and-after study in 1998–2009 in Hainan Province, China (17) found that captive four-eyed turtles *Sacalia quadriocellata* began reproducing after six years after some individuals received hormone injections, but fertility and hatching success of eggs was low. Results were not statistically tested. In 2005–2009, nine of 84 eggs (11%) hatched successfully. In 2004–2008, five of 20 eggs (25%) from hormone injected females were fertile, and 11 of 21 eggs (52%) from females injected with a saline solution were fertile (numbers taken from table). In 2008–2009, three of 43 eggs (7%) from females kept in outdoor pools and given no injections were fertile. In 1998, twenty-eight female and 17 male turtles were acquired and kept in indoor pools (60 x 80 cm). In 2004–2007, eighteen females and 12 males were given luteinizing hormone-releasing hormone analogue (females: 8 μg/kg; males 4 μg/kg) and human chorionic gonadotropin (females: 1600 IU/kg; males 800 IU/kg). Hormones were injected into the hind leg muscles every 10 days up to 10 times/year. The remaining ten females and five males were injected with a saline solution. In 2007–2008, five females and five males were moved to an outdoor pond (10 m^2), and in 2008–2009, eighteen females and 12 males were kept in the outdoor pond.

A study in 1997–2009 in a captive facility in the Seychelles (18a, same experimental set-up as 13a) reported that black mud turtles *Pelusios subniger parietalis* hatched in captivity, but that very few yellow-bellied mud turtles *Pelusios castanoides intergularis* hatched successfully in captivity. In 1997–2009, eighteen black mud turtles and three yellow-bellied mud turtles hatched successfully. The author reported that yellow-bellied mud turtles had a 70% mortality rate during hatching. Captive adult black mud turtles (1–3 males and 3–4 females) and yellow-bellied mud turtles (two males, three females) were held in captivity in 1997–2009 on Silhouette Island. Different pairing approaches were trialled for yellow-bellied mud turtles, including: keeping pairs together, keeping one female with two males, one male with two females, and rotating females between ponds with just males and just females.

A study in 1997–2011 in a captive facility in the Seychelles (18b, same experimental set-up as 13b) reported that Arnold's giant tortoises *Dipsochelys dussumieri arnoldi* and Seychelles giant tortoises *Dipsochelys dussumieri hololissa* bred successfully in captivity. In 2002–2006, forty Seychelles giant tortoises were reared from one female and one male and 140 Arnold's giant tortoises were reared from two females and one male. In 1997–1998, six Seychelles giant tortoises (four males, two females) and six Arnold's giant tortoises (three males, three females) were placed in captivity on Silhouette Island. In 2002, captive groups were reorganised, and all giant tortoises were put together in the same enclosure.

A study in 2001–2009 at captive breeding facilities in Georgia and southern California, USA (19) reported that radiated tortoises *Astrochelys radiata* bred successfully in captivity. In 2001–2009, the captive breeding programmes produced at least 75 juvenile tortoises. Sixty-seven were female and eight were male. Incubation periods for those eggs that hatched in 2006–2009 ranged from 90–120 days. In 2001–2004, tortoises were maintained in a captive breeding facility in Georgia. Tortoises were then moved to a new facility in southern California, where they had access to both indoor and outdoor enclosures. One group of older, wild-caught tortoises were managed to maintain high genetic diversity (details not provided). Another group of captive-born tortoises could choose mates freely. In 2006–2009, eggs were incubated in vermiculite and water at a 2:1 ratio at 28.9°C or 30°C.

A study in 2002–2009 in Florida, USA (20) found that when seasonal variation in temperature and humidity were recreated during incubation of captive Madagascar spider tortoises *Pyxis arachnoides* and flat-tailed tortoises *Pyxis planicauda* eggs, more than half of eggs hatched successfully. In 2002–2009, twenty-six of 50 (52%) spider tortoise eggs and 10 of 10 (100%) flat-tailed tortoise eggs hatched successfully. Of the spider tortoise eggs that failed to hatch, 71% were infertile. There was a large difference between the total incubation period (spider tortoises: 192–303 days; flat-tailed tortoise: 213–275 days) and the length of the incubation period after eggs began to develop (spider tortoises: 82–126 days; flat-tailed tortoise: 73–97 days; see paper for details). Tortoises were acquired in 2002 (numbers not given). Eggs were incubated at 31°C during the day and 26°C at night in vermiculite (1:1 ratio with water) for 8–12 weeks. Eggs were then removed from the incubator and kept at room temperature (20–24°C) for 6–8 weeks, and the vermiculite substrate was left to gradually dry out. Eggs were then returned to the warmer incubation conditions until hatching.

A study in 1999–2011 in a captive breeding facility in Ampijoroa, Madagascar (21) reported that Madagascar big-headed turtles *Erymnochelys madagascariensis* bred in captivity. In total, 94 live hatchlings were produced in three different years (2 in 2004, 52 in 2008, and 40 in 2011). The authors reported that most of the eggs laid were infertile, and that all eggs laid in three nests in 2009–2010 were infertile. The captive breeding programme started in 1999 and from 2011, the captive population comprised six adult males and three adult females. Males and females were put together for the breeding season but kept separately for the rest of the year.

A study in 2001–2013 in Atlanta Zoo, Georgia, USA (22) found that two Arakan forest turtles *Heosemys depressa* bred in captivity and at least one egg hatched from nine of the 11 clutches that were laid. Two captive-bred female Arakan forest turtles laid one clutch/year each for five and six consecutive years respectively. Hatching success ranged between 0–100% for the first female (2–9 eggs laid/clutch) and 13–66% for the second female (4–8 eggs laid/clutch). Of 19 offspring produced, 17 survived in captivity for 1–10 years. One of the adult females bred successfully after three years in captivity, and the second did so during the first year in captivity. The first female died after breeding complications in the sixth year of egg laying. A pair of adult Arakan forest turtles were acquired by Atlanta Zoo in 2001, and a second female was acquired in spring 2009 from Zoo Miami. Adults were maintained in outdoor enclosures during the warmer months of the year and individually indoors during the dry season (see original paper for details).

A study in 2001–2015 in Texas, USA (23) reported that narrow-headed softshell turtles *Chitra indica* produced a single hatchling in captivity. After 14 years, a turtle raised in captivity laid a clutch of 26 eggs. Of the 26 eggs, one (4%) hatched successfully after an incubation period of 69 days, four (15%) completely developed but hatchlings failed to emerge, 10 (39%) failed during development, and 11 (42%) were infertile. The adult female was raised in a 7 x 5 m circular flow-through tank and then moved to a 5 x 2 x 1 m fibreglass tank. No nesting beach was available, and eggs were deposited in the water. Eggs were transferred to a 1:1 mixture of vermiculite and water and incubated at 28°C.

A replicated study in 2004–2016 in captive facilities in the central dry zone of Myanmar (24) found that three captive populations of Burmese star tortoises *Geochelone platynota* bred in captivity. Over 14 years, hatching rates were 50–75% (no further details are provided) and total annual number of hatchlings produced increased from 168 individuals in 2008, to 2,142 individuals in 2016. Female hatchlings that had hatched before 2010 started laying eggs by 2016. The Burmese star tortoise was considered ecologically and functionally extinct in the wild during the 2000s. In 2004, three wildlife sanctuaries located within the tortoises presumed historical geographic range were established as captive assurance colonies, using confiscated juvenile, subadult and adult tortoises and some wild tortoises as the founder population (approximately 175 total tortoises of an equal sex ratio). Tortoises were housed in electric-fenced outdoor enclosures with shelter, food and water provided (see original paper for husbandry details). Nesting activity was monitored and eggs were left in situ to incubate and hatch.

A study in 2012–2017 at Woodland Park Zoo, Washington, USA (25) found that one of two female Indochinese box turtles *Cuora galbinifrons* reproduced successfully in captivity. In 2013–2017, two females produced twelve clutches of 1–3 eggs, with an overall hatching success of 12%. All eggs that hatched came from one female. The average incubation period was 58 days. In 2012, one male and two female turtles were housed separately in glass fronted cages (91 x 135 cm) or concrete enclosures (97 x 183 cm) containing a substrate of soil, mulch and leaf litter, and bark and logs for cover. A water basin was also provided. Ambient temperatures were 25–28°C and humidity was kept at 75–80%. Males were introduced to the female cages for mating purposes. Eggs were moved to a container and suspended over perlite covered with water. A range of temperature regimes were used (see paper for more details), with temperatures ranging from 25.6–29.4°C.

A study in 2013–2017 at the Smithsonian's National Zoological Park, USA (26) found that when incubation temperatures were 26–27°C, one captive female Bourret's box turtle *Cuora bourreti* produced eggs that hatched successfully, whereas at 28–29°C, no eggs hatched. When the incubation temperature was 26–27°C, four of four eggs hatched successfully, with incubation periods of 83–89 days. When the temperature was 28–29°C, zero of 15 eggs hatched successfully, and only three showed any signs of development. Incubation temperatures were 28–29°C in 2013–2016 and 26–27°C in 2017. Eggs were incubated in plastic containers, either partially buried in vermiculite (6:5 ratio with water), suspended over saturated vermiculite, or in the substrate in which they were laid (peat and soil mixture). One female and two males of wild origin were kept in captivity for over 10 years.

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Snakes – Boas and pythons

- **Twelve studies** evaluated the effects of breeding boas and pythons in captivity. Five studies were in the USA[2-4,6,10], two were in the UK[9,12] and one was in each of Jersey[1], Australia[7], India[9] an unknown location[5] and one was a global review[11].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (12 STUDIES)

- **Reproductive success (11 studies):** Five studies in Jersey[1], the USA[2-4] and the UK[9] reported that 1–4 female boas produced litters of 3–34 young, though 2–10 young/litter[1-3] or 38% of young overall[4] were stillborn. One replicated study in the USA[10] reported that a captive breeding programme for ball pythons produced over 5,000 eggs from nearly 800 clutches, with an average hatching success of 81%. Five studies in an unknown location[5], the USA[6], Australia[7], India[8] and the UK[12] reported that female pythons produced clutches of 4–29 eggs, with hatching success of 40–100%[5,6,8,12] or 0–100%[7].

- **Survival (5 studies):** Five studies in the USA[2], Australia[7], India[8] and the UK[9,12] reported that 2–8 captive-bred python hatchlings survived at least two years[7] or 5–8 months[8,9,12], but seven captive-bred emerald tree boas died within three months of birth[2].

- **Condition (1 study):** One global review[11] reported on one study on Jamaica boas that found that captive breeding had a negative effect on genetic variation compared to wild populations.

BEHAVIOUR (0 STUDIES)

A study in 1977 at the Jersey Zoological Park in Jersey (1) reported that two of three female Jamaican boas *Epicrates subflavus* bred successfully in captivity. Two females produced litters of 24 and 34 live young each, with the first litter also containing two stillborn young and an infertile ovum. A third female produced no young. Three female and four male snakes were acquired and housed together in an exhibition enclosure (2.3 x 2.3 x 2.7 m). The substrate was peat and dry leaves; a hollow log and granite boulders were provided; and the air temperature was maintained at 30°C during the day. Young were removed from the main enclosure following birth.

A study in 1973–1977 at the Philadelphia Zoological Garden, USA (2) reported that emerald tree boas *Corallus caninus* bred in captivity in two of four years, though some offspring subsequently died. A female produced six live young and four undeveloped ova one year, and one live young, 10 dead young and three undeveloped ova three years later. Two of six snakes from the first brood died within three months of birth, and the one live snake from the second brood died after three months. An adult pair were received in 1973 and housed together with another female emerald tree boa and a pair of green tree pythons *Morelia viridis* in a fiberglass exhibit (137 x 109 x 239 cm). In 1976, a new captive-bred male emerald tree boa from Fort Worth Zoo was introduced to the female in an exhibit.

A study in 1976–1978 at the New York Zoological Park, USA (3) reported that common anacondas *Eunectes murinus* bred successfully in captivity. In 1978, two females produced 27 and at least 23 live young each, and a third female produced 28 live and two dead young. Two of the females also produced one undeveloped egg. In 1976–1978, three females and one male were housed together in an exhibit (2.5 x 1.9 x 1.5 m) with a substrate of smooth river gravel and a pool of water. Average air temperatures were 27°C, and a heating coil at one end of the cage provided a thermal gradient of 26–30°C.

A study in 1973–1978 in Florida, USA (4) found that three of four Solomon Island ground boas *Candoia carinata paulsoni* reproduced successfully in captivity in at least one of four years. In 1975–1978, four females produced six litters of 16–33 young, though the number of live young/brood varied from 0–100% and in total, 53 of 141 offspring were stillborn. Females reproduced every other breeding season (years taken from table), and one female died after breeding successfully for the second time. One male and two female snakes were acquired in 1973 and two more females were acquired in 1977. Snakes were housed in two glass-fronted wooden cages (90 x 52 x 62 cm; 2 females/cage) with a substrate of ground, dried corn husk. Cages were kept at ambient temperature and humidity during the breeding season.

A study in 1978–1979 in a captive setting [location unknown] (5) found that ball pythons *Python regius* bred successfully in captivity. Two females produced a clutch of eggs each, and two of four and seven of seven eggs hatched successfully. The incubation period for the first clutch was 63 days. Gravid females were moved to an individual aquarium with a substrate of damp peat moss covered with sphagnum moss *Sphagnum* sp., and a piece of driftwood for shelter. Humidity was kept at over 90% and ambient temperatures were 26–30°C and 28–32°C. Eggs were left in the aquarium to be incubated within the female's coils and average coil temperatures were 30.1–30.6°C. One egg that fell outside of the female's coils and was removed and placed in a glass container with damp peat and sphagnum moss. This container was placed back in the aquarium. After two months, one clutch of eggs was removed and incubated at 30.5°C after they stopped adhering to one another and the female was unable to coil around them.

A study in 1978–1979 in the USA (6) found that children's pythons *Liasis childreni* bred successfully in captivity. In 1979, a female produced 10 eggs, six of which hatched successfully. Two eggs failed during development, and two were found to contain fully developed but dead young. Incubation periods were 49–52 days, and the six hatchlings survived at least nine months. A pair of adult snakes was acquired in 1978 and housed in separate 35 litre aquaria with a newspaper substrate and a large piece of bark. In late 1978 to early 1979, they were paired for mating multiple times. When gravid, the female was moved to a large fibreglass cage (91 x 45 x 30 cm) with a thermal gradient. Average temperatures were 28°C during the day and 24°C at night. Eggs were placed on potting soil and incubated at 31°C at 100% humidity.

A study in 1979–1985 in a number of captive settings in Australia (7) found that black-headed pythons *Aspidites melanocephalus*, water pythons *Liasis fuscus*, amethystine pythons *Morelia amethistina* and carpet pythons *Morelia spilota* all reproduced with some success in captivity. Two of three female black-headed pythons produced clutches of eight and 10 eggs, with 100% and 0% respectively hatching successfully. Hatchlings survived at least 24 months. Two female water pythons produced three clutches of 19, 17 and 16 eggs, and 79, 82 and 100% respectively hatched successfully. Further captive females (at least 7) produced clutches of 6–23 eggs (hatching data not provided). An amethystine python produced a clutch of seven eggs, all of which produced live hatchlings (one egg opened artificially). Three carpet pythons produced clutches of 12, 29 and 11 eggs, and 0, 21 and 100% respectively hatched successfully. Snakes were collected and held in captivity and eggs were either removed and incubated in moist vermiculite or were left in situ for the female to incubate (see paper for details).

A study in 2010 in Mini Zoo, Andaman and Nicobar Islands, India (8) reported that a pair of reticulated pythons *Python reticulatus* bred successfully in captivity. One female python produced a clutch of five eggs, two of which hatched successfully. The hatchlings survived for at least five months. In 2010, a pair of reticulated pythons were housed together in a concrete room (3 x 3 x 3 m). After laying, eggs were left to incubate naturally with the female. Hatchlings were measured after hatching, and again after four and five months.

A study in 2008–2010 in a captive setting in Birmingham, UK (9) reported that Madagascar tree boas *Sanzinia madagascariensis* bred successfully in captivity. Two females bred in captivity, with one giving birth to three live young and six infertile eggs, and the second giving birth to five live young, three still-born young and one infertile egg. All eight young snakes survived for at least 6–8 months. In 2008, three tree boas were acquired (two females, one male) and housed individually in enclosures (120 x 60 x 60 cm) with ambient temperatures of 20–28°C and 40–60% humidity. The male was introduced to one female in late 2008–2009, and to the second female in late 2009–2010. All newborn snakes were removed and housed in smaller individual tanks with bark chippings and sphagnum moss *Sphagnum sp*.

A replicated study in 2002–2009 at a commercial breeding company in Utah, USA (10) found that ball pythons *Python regius* bred successfully in captivity. In 2002–2009, a total of 5,344 eggs from 783 clutches were produced, with an average clutch size 7 eggs/clutch, and an average hatching success of 81%. Adult pythons were housed in individual cages (81 x 43 x 18 cm) with a substrate of wood chips. Ambient temperatures were kept between 21–29.5°C year-round, and a hot-spot was available in each cage that was 32°C during the day and 29.5°C at night. Humidity was maintained at 60%. Females (>1,500 g) were placed in cages with males (>500 g) for 1–2 days, and eggs were moved to Styrofoam boxes (29 x 39 x 18 cm) with a glass lid. Eggs were incubated in one-part perlite to two-parts vermiculite (5:1 mixture with water by volume), and temperatures were maintained at 31.4–31.7°C in 2002–2005, and 30.9–31.1°C in 2006–2009.

A review of studies investigating the genetics of captive breeding programmes (11) found that captive breeding reptiles had mixed genetic outcomes in comparison to wild populations. One study found that captive breeding Jamaica boas *Epicrates subflavus* had a negative effect on genetic diversity (measured as expected heterozygosity and number of alleles) compared to wild populations. Two databases (Web of Science and Zoological Record) were searched for studies investigated the genetics of captive populations up until 2010.

A study in 2010 in a captive setting in Birmingham, UK (12) reported that Savu Island pythons *Liasis mackloti savuensis* bred successfully in captivity. One female produced a clutch of nine eggs. Three eggs were infertile, and of the six that were incubated, five hatched successfully. Hatchlings survived for at least eight months. In 2010, one female and three male pythons were acquired. They were housed separately and only introduced to each other for breeding. Ambient temperatures were 29–30°C during the day and 22–25°C at night, and a basking spot at 35°C was provided. Eggs were removed and placed in a plastic box with vermiculite (2:1 mix with water) and incubated at 30°C and 90–100% humidity. Hatchlings were housed individually.

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Snakes – Colubrids

- **Eighteen studies** evaluated the effects of breeding colubrid snakes in captivity. Ten studies were in the USA[1-3,5-7,9,10,12,14], two were the UK[13,18], two were in unknown locations[4,11] and one was in each of Costa Rica[8], Taiwan[15], India[17] and Australia[16].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (18 STUDIES)

- **Reproductive success (18 studies):** Seventeen studies in the USA[1-3,5-7,10,12,14], Costa Rica[8], the UK[13,18], Taiwan[15], Australia[16], India[17] and unknown locations[4,11] reported that 1–2 female colubrid snakes produced 1–12 clutches of 3–16 eggs. Ten of those studies reported hatching success of 67–100%[1,2,4-8,11,15,17], two reported hatching success of 25%[10,13] and two reported that hatching success varied from 0–75%[16,18]. Two of the studies[12,14] reported that at least 18–20 eggs hatched successfully. One study[1] also found that captive-bred offspring produced two clutches of 3–4 eggs and all hatched successfully. One study in the USA[9] reported that three female San Francisco garter snakes produced broods of 9–35 young.
- **Survival (5 studies):** Five studies in the USA[2,9,12] and the UK[13,18] found that 2–20 captive-bred snakes survived for at least 1–3 months[12,13] and 2–3 years[2,18] in captivity, and that from six broods of 9–35 captive-bred San Francisco garter snakes, six young died within four months of birth[9].

BEHAVIOUR (0 STUDIES)

A study in 1968–1976 at Dallas Zoo and Fort Worth Zoo, USA (1) reported that wild-caught trans-pecos ratsnakes *Elaphe subocularis* and their captive-born offspring reproduced successfully in captivity. In 1973, a wild-caught female produced a clutch of six eggs following five years in captivity. Five of six eggs hatched successfully after an incubation period of 105 days. A female from this clutch produced a clutch of three eggs in 1975, and a clutch of four eggs in 1976. All seven eggs hatched successfully following incubation periods of 73–76 days. In 1968, an adult pair of snakes was acquired by Dallas Zoo. Two offspring from this pair were given to Fort Worth Zoo, where they were housed together in a 3-foot fibreglass cage with a pea gravel substrate, rocks and plastic plants. Eggs were removed and incubated in sealed 1 gallon jars in a medium of vermiculite and water (4 oz to one fluid oz water). Jars were opened at two-week intervals to replenish oxygen levels and temperatures were maintained at 28–32°C.

A study in 1974–1976 at Fort Worth Zoological Park, USA (2) reported that Nelson's milksnakes *Lampropeltis triangulum nelsoni* bred successfully in captivity. One female produced one clutch/year for three years and a second female produced a single clutch. Clutch size ranged from 3–5 eggs, and 13 of 16 eggs hatched successfully. Two juveniles that were retained survived at least two years. A male and two females were acquired between 1964–1973 and bred successfully in 1974–1976. All three snakes were housed in a 2-foot fibreglass cage with a substrate of pea gravel. Temperatures fluctuated seasonally between 23–32°C. Groups of 2–3 eggs were transferred to sealed, gallon jars and incubated in vermiculite (1:1 ratio by weight with water). Incubation temperatures were maintained at 23–30°C.

A study in 1973–1977 at Fort Worth Zoological Park, USA (3) reported that Chinese red snakes *Dinodon rufozonatum* bred successfully in captivity in two of four years. In 1973–1976, two females produced clutches of three and seven eggs. Two of three eggs from the first clutch and some from the second (number not provided) hatched successfully. Authors report an incubation period of 49 days. In 1977, a clutch of 12 eggs was produced, but these were still incubating at the time of writing. In 1973, a male and two female snakes were housed together in a 2-foot fibreglass cage, with a gravel substrate, rocks, plants and a water bowl. Temperatures ranged from 23–32°C. Gravid females were moved to a separate cage and provided a bowl containing damp vermiculite and/or moss. Eggs were placed in a 5-gallon aquarium in a 1:1 mixture of vermiculite and water, and incubated at 29–35°C.

A study in 1974–1976 in a captive setting [location unknown] (4) reported that gray-banded kingsnakes *Lampropeltis Mexicana alterna* bred successfully in captivity. In 1976, a female produced a clutch of eight eggs, seven of which hatched successfully after an incubation period of around 70 days. The 8th egg was infertile. In 1974–1975, an adult pair of snakes was acquired. They were housed separately in 10 gallon tanks with a newspaper substrate, and temperatures were maintained at 24–28°C. In 1976, the female was introduced to the male on three consecutive days and was then moved to a 5.5 gallon aquarium half filled with damp potting soil. Eggs were removed and placed in a plastic box with vermiculite and water, and incubated at 28–32°C.

A study in 1971 in the USA (5) reported that scarlet kingsnakes *Lampropeltis triangulum elapsoides* bred successfully in captivity. One female produced a clutch of four eggs, and all four hatched successfully after an incubation period of around 66 days. In 1971, a pair of adult snakes was acquired and housed in separate 38 litre aquaria with paper towel substrates and pine bark. Temperatures ranged from 21–28°C. The male was transferred to the female's aquaria for mating purposes. Eggs were incubated at 25–31°C between layers of damp paper towels in a 1 litre plastic container, sealed with clear plastic wrap. All snakes were released around two months after hatching.

A study in 1979–1981 at Memphis Zoo and Aquarium, USA (6) reported that black pine snakes *Pituophis melanoleucus lodingi* bred successfully in captivity. In 1981, a single female produced seven eggs, and all seven hatched successfully. In 1979–1980, two female snakes and one male were obtained from Alabama and housed individually in 113 litre aquaria, with temperatures ranging from 25–32°C in summer and 9–18°C in winter. In March 1980 and April 1981, the first female was paired with the male, but no mating activity was observed. In April 1981, the second female was paired with the male and they mated successfully. Eggs were transferred to an 11 litre, sealed plastic box with small holes drilled in the sides and incubated in vermiculite (1:1 ratio by volume with water). Incubation temperatures were maintained at 24–31°C but dropped as low as 6°C at night.

A study in 1981–1982 at Cheyenne Mountain Zoo, USA (7) reported that a pair of Great Basin gopher snakes *Pituophis melanoleucus deserticola* bred successfully in captivity. In 1982, the female produced a clutch of seven eggs, and all seven hatched successfully after a 55–58-day incubation period. In 1981, a pair of snakes were acquired, and in 1982 they were placed together in an aquarium (73 x 31 x 28 cm) with a gravel substrate. A plastic container containing damp vermiculite was placed in the aquarium but was ignored by the female. Eggs were collected, wiped clean with zephiran chloride (1:750 solution) and covered with damp vermiculite in a stainless-steel container. They were incubated at 29.5°C.

A study in 1982–1983 in San José Province, Costa Rica (8) reported that a pair of mussuranas *Clelia clelia* bred successfully in captivity. The female produced a clutch of 10 eggs, seven of which hatched successfully after 117–120 days of incubation. One egg did not develop, and two eggs contained fully developed but dead young with some physical deformities. In 1982, a pair of snakes were housed together in a wooden cage (122 x 62 x 60 cm). Temperature was 25°C and humidity was 60%. Eggs were incubated at 26–28°C in a fiberglass case (21 x 21 cm) on damp cotton. The case was kept inside a plastic bag and was opened daily for ventilation.

A study in 1983–1986 at Dallas Zoo and Fort Worth Zoo, USA (9) reported that San Francisco garter snakes *Thamnophis sirtalis tetrataenia* bred successfully in captivity. In 1984–1986, three females produced six broods of 9–35 young/brood, following gestation periods of 79–98 days. Six young died or had to be put down within four months of birth. The male to female sex ratio of broods ranged from 19:16 to 5:12. Snakes were housed in plastic show boxes, 1 gallon glass jars or 5 gallon aquaria (36 x 22 x 26 cm) with a paper substrate and plastic hide boxes, bark and plastic leaves. Ambient temperatures were 27–30°C at Dallas Zoo and 21–32°C at Fort Worth Zoo, and spotlights provided basking spots at 32°C for gravid females.

A study in 1986–1987 in the USA (10) reported that Louisiana pine snakes *Pituophis melanoleucus ruthveni* produced a single hatchling in captivity over two years. In 1986, a female produced a clutch of two infertile eggs. In 1987, the same female produced a clutch of four eggs, one of which hatched successfully and three of which did not develop. In 1986, a female and two male snakes were acquired and housed separately in 114 litre aquaria with a substrate of wood shavings. Temperatures ranged from 23–32°C. In March–May, snakes were introduced to each other for mating. Eggs were moved to an 11 litre plastic box containing moist vermiculite and incubated at 25–31°C. The boxes had small holes drilled in the sides and were opened for a few seconds every week.

A study in 1989–1995 [location unknown] (11) reported that Mandarin rat snakes *Elaphe mandarina* bred successfully in captivity. In 1993–1995, a female produced three clutches (at least 5, 6 and 6 eggs/clutch), with 16 eggs hatching successfully. Incubation periods were around 48 days, and the ratio of males to females was 2:1. In 1989, a pair of snakes (recently captive-born) was acquired and housed together in a glass enclosure (40 x 25 x 25 cm). Temperatures were maintained at 22–26°C during the day and 16–18°C at night, and humidity was high. Snakes were then moved to separate enclosures (60 x 40 x 40 cm). The snakes reached maturity in 1992, and in 1993, the female was introduced to the male's enclosure. Eggs were removed and incubated at 25–28°C in very high humidity on sphagnum moss *Sphagnum* sp..

A study in 1993–1996 at the Riverbanks Zoo, South Carolina, USA (12) reported that Oates' twig snakes *Thelotornis capensis oatesii* bred successfully in captivity. A single female produced two clutches/year of 3–11 fertile eggs/clutch, and at least 20 eggs hatched successfully. Incubation periods ranged from 59–61 days (at 28.9°C) to 72–76 days (at 26.7°C). All hatchlings survived for at least three months. Authors also reported that in 1997, two captive bred females produced seven and five fertile eggs each at St. Louis Zoo, Missouri. In 1993–1994, one wild female and three wild males were acquired and subsequently paired up in a 122 x 107 x 81 cm tank, with basking spots between 29–35°C. Following mating, the female was moved to a smaller tank (61 x 41 x 31 cm). Eggs were incubated in vermiculite (2:1 ratio with water) in a 0.5 litre covered glass jar, and incubation temperatures ranged from 26.7–28.9°C.

A study in 2007–2009 in a captive setting in Birmingham, UK (13) reported that a pair of red-tailed ratsnakes *Gonyosoma oxycephala* bred successfully during one of two years. In 2007–2008, a female produced four infertile eggs. In 2008–2009, the same female produced a clutch of four eggs, none of which hatched successfully (embryos died during development), and a clutch of four eggs, three of which hatched successfully. All three hatchlings survived for at least a month. In 2007, a pair of ratsnakes were acquired and housed in individual enclosures and only introduced to each other for mating. Ambient temperatures were 25–32°C during the day and 18–20°C at night. Eggs were removed and placed in a plastic container, partially buried in vermiculite (2:1 mix with water) and covered in damp sphagnum moss *Sphagnum* sp. and incubated at 30°C. The container was opened every two days.

A study in 2010 in Missouri, USA (14) reported that red cornsnakes *Pantherophis guttatus* bred successfully in captivity. Two females produced at least 18 hatchlings. All captive offspring came from a single male and two female snakes. Eggs were moved to an incubator where the temperature was 28°C and the humidity was ≤80%.

A study in 2011–2013 in Taiwan (15) reported that a pair of Indo-Chinese rat snakes *Ptyas korros* reproduced successfully in captivity. In 2013, one female produced a clutch of six eggs, four of which hatched successfully. One egg was infertile, and one contained twin snakes that died before hatching. In 2011, a pair of rat snakes were brought into captivity. Eggs were placed in an incubator, where temperatures varied from 28°C during the day and 24°C at night.

A study in 1984–1997 at Taronga Zoo, Sydney, Australia (16) reported that brown tree snakes *Boiga irregularis* reproduced successfully after four years in captivity. In 1988–1995, a female produced seven clutches of 10–16 eggs. Authors report hatching data for three clutches, with nine of 12 (75%), zero of 11 (0%) and

13 (clutch size unknown) eggs hatching successfully. Reported incubation periods ranged from 82–92 days. Five of six eggs hatched successfully from an additional clutch that was laid soon after the snakes arrived in captivity, and authors reported that the female was most likely gravid when captured. A pair of snakes were acquired in 1984, and authors reported that incubation was attempted for four clutches of eggs. Details on incubation conditions were not reported.

A study (year not provided) in a captive setting in Pilikula Biological Park, Karnataka, India (17) reported that montane trinket snakes *Coelognathus helena monticollaris* bred successfully in captivity. Two females laid one clutch each of eight and 12 eggs respectively, and 100% of eggs hatched successfully. Two pairs of adult montane trinket snakes were acquired and housed in one enclosure (2 x 2 m) with a soil and leaf litter substrate, along with some plants, deadwood, stones and a water pit. Temperatures were maintained at 22–28°C, and humidity was 80–90%. Eggs were removed and incubated in a plastic box with a soil substrate at 25–28°C and 80–90% humidity.

A study in 2008 and 2013–2016 at London Zoo, UK (18) reported that two pairs of rhino rat snakes *Gonyosoma boulengeri* bred successfully in captivity. Two females laid one clutch each of nine eggs (including one infertile egg from one female). Three eggs from one of the clutches (33%) and six from the other (66%) hatched successfully. At least three of the hatchling snakes survived for at least three years. One pair of snakes was acquired in 2008, and a second pair in 2013. The 2013 pair was housed in an enclosure with a chipped bark substrate, a range of different plants and branches, and a hide box containing damp sphagnum moss *Sphagnum* sp. Ambient temperatures ranged from 24–28°C in summer and 18–26°C over winter, and a basking spot at 30–34°C was also provided. Eggs were removed and placed in vermiculite (1:1 mix with water by weight) and incubated at 28°C. Hatchlings were moved to individual plastic tanks.

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Snakes – Elapids

- **Four studies** evaluated the effects of breeding elapid snakes in captivity. Three studies were in Australia[1-3] and one was in India[4].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (4 STUDIES)

- **Reproductive success (4 studies):** Three studies in Australia[1,2] and India[4] reported that 1–4 females elapid snakes produced clutches of eggs in captivity, with 26–93% hatching successfully. One study in Australia[3] reported that two generations of death adders produced litters of 17–25 young in captivity, though 20 were still born.
- **Survival (2 studies):** Two studies in Australia[1] and India[4] reported that two western brown hatchlings survived 2–3 years[1] and 87% of king cobra hatchlings survived one year in captivity[4].
- **Condition (1 study):** One study in Australia[1] reported that eight of 15 captive-bred western brown snake hatchlings lacked one or both eyes.

BEHAVIOUR (0 STUDIES)

OTHER (1 STUDY)

- **Offspring sex ratio (1 study):** One study in Australia[3] reported that 55% of captive-bred Australian death adders were female.

A study in 1971–1979 at The Royal Melbourne Zoo, Australia (1) reported that western brown snakes *Pseudonaja nuchalis* and eastern brown snakes *Pseudonaja textillis* bred successfully in captivity. One western brown female produced 37 eggs over three years, 15 of which hatched successfully. Eight hatchlings were lacking one or both eyes, and two that were retained survived at least 2–3 years. Three eastern brown females produced 69 eggs over five years, 18 of which hatched successfully. Incubation duration for western browns was 52 days at 30°C and 59–64 days at 28°C, and for eastern browns it was 40–42 days at 30–31°C and 49–52 days at 28°C. Newly laid eggs were collected and placed in vermiculite-filled containers for incubation.

A study in 1973–1982 in Australia (2) reported that Collett's snakes *Pseudechis colletti* bred successfully in captivity. One female produced five clutches in seven years, and two other females produced one clutch each in one year (7–14 eggs/clutch). Five of seven (71%) and 13 of 14 (93%) eggs from the latter two clutches hatched successfully. No hatching data is reported for the five clutches produced by one female. Incubation periods ranged from 56–70 days. Three female and four male snakes were acquired during the 1970s and early 1980s and placed together in pairs at ambient temperatures. Eggs were incubated in damp vermiculite (50% water by weight) at temperatures of 27–30°C or 29.5–32°C.

A study in 1971–1977 in New South Wales, Australia (3) reported that two generations of Australian death adders *Acanthophis antarcticus* bred successfully in captivity. For three years the wild-caught pair did not reproduce, but in 1971–1977, the snakes and their offspring produced 12 litters of 17–25 young. At least four litters came from the wild-caught snakes, and at least four came from their captive-born offspring. A total of 20 young from three litters were stillborn, and five undeveloped eggs were produced. Seven of eight litters that were sexed at birth had sex ratios skewed slightly towards females, with females making up 55% of young. In 1964–1966, a pair of adult death adders were acquired and after the original male died in 1971, a new male was introduced. Adders were housed in various aquaria (60 x 30 cm to 150 x 45 cm in size), with a substrate of gravel and leaf litter. All aquaria were exposed to ambient temperatures.

A study in 1996 at Madras Crocodile Bank, India (4) reported that three of four female king cobras *Ophiophagus hannah* laid eggs that hatched and survived in captivity. All four captive female king cobras laid a single clutch (16–37 eggs/clutch). In one clutch, none of the eggs were viable (0 of 18 eggs). In the remaining three clutches, 75–100% of eggs were viable (clutch a: 18 of 18 eggs viable, clutch b: 34 of 37 eggs viable, clutch c: 12 of 16 eggs viable). Hatching success of viable eggs ranged from 41–61% (clutch a: 11 of 18 eggs hatched, clutch b: 14 of 24 eggs hatched, clutch c: 5 of 12 eggs hatched). Twenty-six of 30 hatchlings survived at least one year in captivity. In 1996, Madras Crocodile Bank acquired four female and three male adult king cobras from Indian zoos or from government seizures from snake collectors. Snakes were housed in indoor enclosures (see original paper for husbandry details). Eggs were incubated individually on damp vermiculite substrate (1:0.8 vermiculite: water by weight) at temperatures of 27.5–33°C. Egg viability and hatching success was evaluated after emergence.

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Snakes – Vipers

- **Thirteen studies** evaluated the effects of breeding vipers in captivity. Nine studies were in the USA[1-3,5-10], three were in unknown locations[4,11,12] and one was in Columbia[13].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (13 STUDIES)

- **Reproductive success (13 studies):** Thirteen studies in the USA[1-3,5-10], Columbia[13] and unknown locations[4,11,12] reported that 1–4 female vipers, including three captive-bred offsping[2], produced litters of 1–18 live young[1,3-8,10-12] or clutches of 1–26 eggs with hatching success of 63–81%[2,9,13]. One study[13] also reported that none of three Chocoan bushmaster eggs that were removed and incubated artificially fully developed.
- **Survival (5 studies):** Three studies in the USA[1,2,5] and one in an unknown location[12] reported that of 10–49 captive-bred young snakes, 1–9 died soon after birth or within three months. One study[12] also reported that one pair of adult adders died shortly after arriving in captivity. One study in an unknown location[11] reported that four captive-bred Radde's vipers survived for at least eight months
- **Condition (1 study):** One study in an unknown location[12] reported that two of 10 captive-bred Nikolsky's adders had some physical deformities.

BEHAVIOUR (0 STUDIES)

OTHER (2 STUDIES)

- **Offspring sex ratio (2 studies):** Two studies in the USA[1] and an unknown location[4] reported that the sex ratio of captive-bred lower California rattlesnakes was 2:12[1] and Russell's vipers was 8:6[4] females to males.

A study in 1972–1976 at Fort Worth Zoological Park, and a second private collection, USA (1) reported that lower California rattlesnakes *Crotalus enyo enyo* bred successfully in captivity. In 1974–1976, three females produced four litters of 2–7 young each following gestation periods of 176–299 days. One litter of seven included two young that were stillborn and one that died soon after birth. The overall sex ratio was 12 males to two females and one of unknown sex. In 1972–1974, one female and one male snake were acquired by Fort Worth Zoo, and two females and a male were acquired by a second private collection. Snakes were housed in a 60 cm fibreglass exhibit with a gravel substrate, or a 38 litre tank with a newspaper substrate. Temperatures ranged from 22–33°C.

A study in 1969–1978 at Columbus Zoo, Ohio, USA (2) reported that two generations of Palestine saw-scaled vipers *Echis coloratus* bred successfully in captivity. After not reproducing for two years, a female produced seven clutches of 5–12 eggs over seven years. Three females from the first of these clutches (hatched in 1971) went on to produce a total of four clutches of 1–9 eggs over three years. In total, 49 of 72 eggs hatched successfully, with an average hatching success/clutch of 64%, though three hatchlings died within 24 hours of emergence. Six clutches also contained 1–3 infertile egg masses. In 1964–1968, a pair of snakes was acquired and in 1969 they were placed together. The original pair were housed in a display cage (68 x 56 x 52 cm), and offspring were housed individually and then combined into groups during the spring in aquaria of various sizes.

A study in 1973–1978 in Texas, USA (3) reported that a pair of lancehead rattlesnakes *Crotalus polystictus* bred successfully after five years in captivity. In 1978, the female gave birth to a litter of three young. The pair of snakes was brought into captivity in 1973 and housed in a glass fronted display (60 x 40 x 55 cm) with a substrate of small rocks. Cover was provided by larger rocks, leaves and dried plants. Temperatures were maintained at 29°C.

A replicated study in 1972–1978 [location unknown] (4) found that Russell's vipers *Vipera russelli* bred successfully in captivity. In 1972–1978, eight females each produced one litter of 9–22 live young. The average sex ratio of litters was eight females for every six males. Five pregnant vipers were imported from Pakistan (origin unclear) and the other three females were captive born. Snakes were kept in large terraria (3.8 x 2.3 x 1.1 m) and smaller cages (0.5–1.0 x 0.5 x 0.5 m), with a thermal gradient of 18–50°C and minimum temperature of 18°C at night.

A study in 1973–1979 at the Oklahoma City Zoo, USA (5) found that Mexican cantils *Agkistrodon bilineatus* bred successfully in captivity. In 1973, two females produced one litter each of six and 12 live young, though six young died shortly after birth. In 1977, one female produced a litter of nine young, two of which were still alive after three months, and one of which survived at least 2 years. In 1966–1973, a male and two females were acquired. They were housed together from 1973 in a glass-fronted wooden enclosure (79 x 61 x 46 cm) with a gravel substrate, rocks and plastic vegetation. The male died following removal of a tumour in December 1976.

A study in 1970–1980 at the Houston Zoological Gardens, USA (6) found that Uracoan rattlesnakes *Crotalus vegrandis* bred successfully in captivity. In 1976–1980, two females produced five litters of 2–8 young. In 1970, a gravid female was brought into captivity and produced a litter of three young. In 1970, two females and a male were brought into captivity. They were housed in either an exhibit (80 x 90 x 140 cm) with a substrate of gravel, rocks and plastic plants, or aquaria of various sizes with a paper substrate and hide boxes. Temperature was maintained at 28°C and humidity at 75%. Adult snakes were occasionally temporarily separated and then returned to the same enclosure.

A study in 1969–1980 at Houston Zoological Gardens, USA (7) found that Aruba Island rattlesnakes *Crotalus unicolor* bred successfully in captivity. In 1973–1980, four females produced a total of nine litters of 2–5 live young/litter, or 3–8/litter when including still born young and infertile eggs. Three of these litters (2–4 live young/litter) were produced by two females born in captivity. The oldest female also produced a single infertile egg on three occasions. In 1969–1976, two female and two male snakes were acquired. Snakes were housed in a glass fronted enclosure (80 x 90 x 140 cm) with a gravel substrate, rocks and plastic plants, or in aquaria of various sizes with a paper substrate and hide boxes. Temperatures were maintained at 28°C and humidity was 85% in the enclosure and 75% in the aquaria. Snakes were occasionally temporarily separated and then returned to the same enclosure.

A study in 1981–1982 at Fort Worth Zoological Park, USA (8) found that eyelash vipers *Bothrops schlegeli* bred successfully in captivity. In 1981–1982, three females produced a total of five litters of 6–17 young/litter. In 1981, four captive born snakes (3 females, 1 male) and one of unknown origin (male) were acquired and housed in either a 91 cm fibreglass enclosure with a gravel substrate or in glass aquaria with a newspaper substrate. Rocks, plastic plants, branches and a water bowl were provided. Temperatures were maintained at 24.4–29.4°C. Offspring were removed and placed in plastic containers of various sizes.

A study in 1978–1982 at Dallas Zoo, USA (9) found that Cretan vipers *Vipera lebetina schweizeri* bred successfully in captivity. In 1980–1982, two years after a male and female were first housed together, the female produced 26 eggs, 21 of which hatched successfully. The incubation period ranged from 37–48 days. An adult pair of Cretan vipers were housed in a fibreglass enclosure (78 x 60 x 30 cm) with a sand and gravel substrate that was maintained at a temperature of 26–29°C. A container filled with moist sphagnum *Sphagnum* sp. was provided for egg laying, and eggs were collected and incubated at 27–31°C.

A study in 1985–1986 in the USA (10) found that eyelash vipers *Bothrops schlegeli* bred successfully in captivity. In 1986, two females gave birth to 15 and 18 live young each following gestation periods of at least 237 and 338 days. In 1985, two females were placed with a male in a glass aquarium (60 x 40 x 32 cm). The male was removed one month later. Temperatures varied from 19–35°C under a heat lamp each day.

A study in 1989–1991 [location unknown] (11) found that two captive female Radde's vipers *Vipera raddei raddei* bred successfully in captivity. In 1991, one female gave birth to one live young and three infertile egg masses, and the other female gave birth to three young and two infertile egg masses. The young snakes survived for at least eight months. Adult snakes were placed together in a 140 x 60 x 60 cm terrarium, with a substrate of fine gravel, larger rocks, and shelters made from bark and plywood. The terrarium was kept at a temperature of around 25–32°C.

A study in 1999–2003 [location unknown] (12) found that Nikolsky's adders *Vipera nikolskii* reproduced successfully in captivity in one of two years. In 2002, one female produced five infertile egg masses. In 2003, one female produced seven live young, and a second female produced three live young and four infertile egg masses. Two of the young snakes had physical deformities and a third died shortly after emergence. One pair of adult snakes died soon after arriving in captivity. In 1999–2000, three pairs of snakes were acquired from Ukraine. Snakes were housed with a newspaper substrate and plastic hide box, and temperatures were maintained at 22–28°C during the day with a basking area at 35°C, and 18–22°C at night. Juvenile snakes were moved in groups of 2–3 to small containers with a paper substrate, where temperatures were maintained at 22–25°C during the day and 20–22°C at night.

A study in 2007–2013 in Medellín, Colombia (13) found that one pair of Chocoan bushmasters *Lachesis acrochorda* bred successfully in captivity. A female produced a clutch of 11 eggs, seven of which hatched successfully. Incubation periods ranged from 93–96 days. None of three eggs that were removed and incubated artificially fully developed. In 2007–2013, two female and two male snakes were held together in an enclosure (3.6 x 2.5 x 2.7 m) with a substrate of gravel, soil and rice husks, with a water source, larger rocks and plants. Temperatures were maintained at 19–28°C and humidity at 75–95%. In February

2013, just prior to the breeding event, humidity was increased to >85%. Three eggs were removed and incubated in a 1:1 mix of vermiculite and water at 24.6–25°C and 77–80% humidity. The remaining eight eggs were left in the enclosure for 74 days, at which point they were moved to an incubator, with temperatures of 24.5–27.7°C and 78–91% humidity.

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Lizards

- **Twenty-three studies** evaluated the effects of breeding lizards in captivity. Ten studies were in the USA[1,3,5,6,9,11,13,14,16,17], three were in Australia[18,22,23], two were in the UK[15,19] and one was in each of Switzerland[2], an unknown location[4], the Arabian Peninsula[7], Mexico[8], Italy[10], Spain[12], Bahamas[20] and Jamaica and the USA[21].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (23 STUDIES)

- **Abundance (2 studies):** One replicated study in Spain[12] reported that a captive-breeding programme for large psammodromus lizards produced 365 juveniles for release over two years. One replicated study in Australia[22] reported that captive populations of Lister's geckos and Christmas Island blue-tailed skinks at two facilities grew or remained stable over 4–5 years.

- **Reproductive success (22 studies):** Eighteen studies (including seven replicated studies) in the USA[1,3,5,6,11,13,16,17], Switzerland[2], an unknown location[4], the Arabian Peninsula[7,] Mexico[8], Italy[10,] Spain[12] the UK[15,19] and Australia[18,23] reported that captive lizards produced one or more clutches of 2–21 eggs[1,3,4,6,7,8,10,12,13,15-19,23], 3–12 eggs/year[11] or gave birth to 21 live young[2]. Eleven of the studies[1,7,8,10,12,13,15-17,19,23] reported hatching success of 45–96%. Three of the studies[3,4,6] reported hatching success of 0–40%[4], 0–43%[3] or 0–100%[6]. One of the studies reported hatching success of <10%[5]. One of the studies[23] also found that hatching success for Australian painted dragon eggs was similar across all incubation temperatures used, but higher for eggs laid earlier in the season. One of two studies (including one replicated study) in Jamaica and the USA[21] and the Bahamas[20] reported that captive breeding programmes lasting 19 and 24 years produced 73 and five Jamaican iguana hatchlings respectively[21]. The other study[20] reported that over 2.5 years, captive San Salvador rock iguanas produced only a single hatchling. One controlled study in the USA[14] found that captive-reared western fence lizard females housed individually or in pairs produced more clutches with fewer infertile eggs compared to females kept in groups of four or eight. One replicated, before-and-after study in the USA[9] found that curious skinks kept in smaller breeding groups and provided nutrient rich food produced more clutches of eggs than skinks that were kept in larger groups and given regular food.

- **Survival (9 studies):** Seven studies (including four replicated studies) in an unknown location[4], Mexico[8], Italy[10], the USA[11,13,17] and the UK[19] reported that 4–23 captive-bred lizards, or some individuals[11,19], survived for six weeks[13] or at least six months to three years[4,8,17,19], or that individuals of three species survived to reach adult size[10]. Two studies in the USA[1,5] reported that one of three[1] and eight of 10[5] captive-bred lizards died within one day[1] or 18 months[5].
- **Condition (1 study):** One controlled study in the USA[16] reported that giant horned lizard eggs incubated at 26.5°C produced larger hatchlings compared to those incubated at 28°C.

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One study in the USA[17] reported that captive female Yuman fringe-toed lizards selected an 8:1 sand:water mixture when laying eggs.

A study in 1970–1979 at the Houston Zoological Gardens, USA (1) reported that Gila monsters *Heloderma suspectum* bred in captivity and one of two females produced young that survived beyond hatching. In 1979, one female produced four eggs, two of which hatched successfully, though one hatchling was removed from the egg by hand. A second female produced two eggs, one of which hatched successfully, though the hatchling died after one day. Incubation periods ranged from 112–126 days. The remaining three eggs did not develop successfully and were discarded. Five Gila monsters were acquired in 1970–1978, including two females, one male, and two of unknown sex. The lizards were kept together in a glass fronted exhibit (130 x 88 x 140 cm). The average temperature was 28°C, and the lizards were exposed to a UV lamp for 15–20 minutes/day. Eggs were moved to a plastic container (33 x 16 x 8 cm) and incubated in vermiculite (4:1 mixture with water) at 28° C.

A study in 1973–1984 at Zurich Zoo, Switzerland (2) reported that prehensile-tailed skinks *Corucia zebrata* bred successfully in captivity. In 1973–1984, twenty-one skinks were born in captivity, though seven were stillborn. Two young skinks were cannibalised by adults. Skinks were acquired in 1973–1975 from the Bougainville Islands and Solomon Islands. After losing young skinks to cannibalism, gravid females were moved to separate terraria (100 x 50 x 90 cm) containing leaves, peat moss, branches, bark and hollow logs.

A study in 1974–1979 in Arizona, USA (3) reported that two species of giant chuckwalla *Sauromalus varius* and *Sauromalus hispidus* bred successfully after 2–3 years in captivity. In 1977–1979, two female *Sauromalus varius* and one female *Sauromalus hispidus* produced a total of 90 hatchlings from seven broods (5–23 hatchlings/brood). Three of seven clutches of eggs were discovered (19, 21 and 22 eggs/clutch), and the average hatching success was 43% (four clutches remained undiscovered). Five broods were *Sauromalus varius* (6–23 hatchlings/brood), one was *Sauromalus hispidus* (5 hatchlings/brood), and one was a hybrid of both species (12 hatchlings/brood). One clutch of 18 undeveloped eggs was discovered that was likely laid in 1976, and 15 eggs that were artificially incubated all failed to hatch. In 1974–1975, two pairs of *Sauromalus varius* and one pair of *Sauromalus hispidus* were acquired and housed in an outdoor enclosure (670 x 580 x 120 cm) with natural clay soil covered with 46 cm of desert wash sand. Temperatures ranged from -1–45°C. Fifteen eggs from a clutch discovered in 1977 were removed an incubated at 32°C. Hatchlings were moved to indoor terraria (200 x 100 x 70 cm), with 10 hatchlings/terrarium and temperatures of 25–38°C.

A study in 1980–1986 [location unknown] (4) found that Timor Monitors *Varanus timorensis similis* bred successfully in captivity and produced hatchlings in two of three years. A female produced one clutch/year of five eggs for three years. Zero eggs hatched from the first clutch, and two eggs from both subsequent clutches hatched successfully (40% hatching success). Incubation periods ranged from 121–140 days, and the four hatchlings survived for at least 2–3 years. In 1980–1981, a wild-caught female and three male monitors were housed in a glass fronted terrarium (195 x 36 x 60 cm) with a newspaper substrate, branches and rocks. Temperatures were 26–30°C during the day and 20–23°C at night. Eggs were moved to a 40 litre aquarium containing 1 cm of water and placed on a raised aluminium screen. The temperature was maintained at 28–30°C, and humidity was 90–100%. Hatchlings were housed individually in 60 litre aquaria with a newspaper substrate and cardboard tubes.

A replicated study in 1990–1991 in Texas, USA (5) found that captive-reared Malagasy panther chameleons *Chamaeleo pardalis* had low breeding success in captivity. Ten females each produced 1–5 fertile clutches of eggs, and while all fertile eggs developed to full-term young, <10% hatched successfully. Eight of 10 females died within 18 months. Chameleons were housed in fishbowls (1 litre) or aquaria of increasing sizes (8, 13 then 50 litre) as they grew larger. Dead twigs were provided, and temperatures ranged from 30–33°C during the day and 19–23°C at night.

A study in 1992–1993 at Dallas Zoo, USA (6) found that Gould's monitors *Varanus gouldii* and Gray's monitors *Varanus olivaceus* bred successfully in captivity. In 1992–1993, a female Gould's monitor produced two clutches/year of 5–10 eggs/clutch. One clutch of five eggs was infertile and hatching success of the other clutches ranged from 50–100%. A female Gray's monitor produced two clutches/year of 4–8 eggs/clutch, though only a single egg from the first clutch hatched successfully. Females were introduced to a male regularly for several days until courtship or copulation was observed.

A study in 1992–1993 on the Arabian Peninsula (7) found that a pair of blue-tailed Oman lizards *Omanosaura cyanura* bred successfully in captivity. In 1993, one female laid five clutches of three eggs and one clutch of 2 eggs. Overall, five of 11 eggs (45%) hatched successfully, and no hatching information was available for two clutches of three eggs each. In 1992–1993, a female and two male lizards were brought into captivity and initially housed together until one male was removed. The terrarium (90 x 40 x 60 cm) contained sand, stones and an *Aloe* sp. The sand was maintained at 30°C, and during the day temperatures were increased to 60°C for several hours using a spotlight. Eggs were removed and incubated at 28°C.

A study in 1985–1995 in Mexico City, Mexico (8) found that a pair of Mexican acaltetepons *Heloderma horridum* bred successfully in one of nine years in captivity. In 1994, a female produced a clutch of 11 eggs, nine of which hatched successfully. Four hatchlings died after two months and five survived for at least one year. Mating activity in three of the previous eight years produced two clutches of one and 15 eggs, none of which developed. An adult pair of lizards were acquired in 1985. Eggs were removed and incubated at 29°C and 95–100% humidity in plastic boxes that were opened daily.

A replicated, before-and-after study in 1995–1997 in Texas, USA (9) found that captive curious skinks *Carlia ailanpalai* produced more clutches of eggs when housed in smaller breeding groups and fed with nutrient rich crickets compared to when they were housed as a single group and fed normal crickets. Results were not statistically tested. When skinks were housed as smaller breeding groups (14 lizards in 4 aquaria and 13 lizards in 3 aquaria) and fed nutrient rich crickets, 76 clutches were produced over 7 months and 16 clutches were produced over 4 months. All clutches from these smaller breeding groups hatched successfully, though hatchlings from one clutch had physical deformities. When 14 skinks were housed in a single aquarium and fed with normal crickets, three clutches of eggs were produced in nine months, one of which hatched successfully. In 1995, eight female and 6 male skinks were received from Guam, Mariana Islands. Skinks were housed in 75 litre aquaria with a substrate of sand or sand and potting soil. Temperatures ranged from 20–32°C, though basking spots at 55°C were available in the aquaria used for smaller breeding groups. Humidity ranged from 60–90%. Nutrient rich crickets were created by feeding crickets with powdered T-Rex® Calcium Plus cricket food.

A replicated study in 1997–2000 in Italy (10) found that successful captive breeding was achieved for lined flat-tail gecko *Uroplatus lineatus*, day gecko *Phelsuma madagascariensis* and Standing's day gecko *Phelsuma standingi.* Lined flat-tail gecko (4 clutches of 2 eggs/female/year, 93% hatching success, 100% survival to adult), day gecko (8 clutches of 2 eggs/female/year, 95% hatched, 100% survival) and Standing's day gecko (8 clutches of 2 eggs/female/year, 93% hatched, 100% survived) bred successfully in captivity. The following species laid eggs in captivity, but no information on hatching success was available: Parson's chameleon *Calumma parsonii* (1 clutch of 40 eggs/female/year), panther chameleon *Furcifer pardalis* (6 clutches of 30–50 eggs/female/year) and satanic leaf-tailed gecko *Uroplatus phantasticus* (3–4 clutches of 2 eggs/female/year, 2). The estimated cost for one captive-bred individual was: €44.50 for either day gecko species, €60.00 for leaf-tailed or flat-tailed geckos and €6.30 for panther chameleons. Animals were imported from Madagascar in 1997–1998 or were obtained from private breeders or other facilities. Reproduction was monitored in captivity over two years. Some data were obtained from private breeders. Costs were calculated for Italy.

A replicated study in 1996–2001 in Florida, USA (11) found that seven species of dwarf gecko *Sphaerodactylus* spp. reproduced successfully in captivity. Twenty-six females from seven species produced an average of 3–9 eggs/year in their first year, and nine females from four species produced 5–12 eggs/year in their second year. Average incubation time for 74 clutches from seven species ranged from 63–86 days. Most individuals of all seven species survived for at least two years. In 1995–2001, geckos were acquired and housed individually or in male-female pairs in plastic containers (2 litres) with a substrate of soil and peat moss, small rocks, leaves and pine bark. Temperatures ranged from 24–33°C in April–October, and 14–23°C in November–March. Eggs were moved to individual plastic containers and placed on sterile soil and peat moss. Incubating eggs were exposed to the same seasonal variations in temperature.

A replicated study in 2001–2002 in a laboratory in northern Spain (12) found that large psammodromus lizards *Psammodromus algirus* bred successfully in captivity. Hatching success (2001: 92%; 2002: 87%) and hatchling survival (2001: 91%; 2002: 83%) were high in both years, yielding 178 and 187 juveniles for release in 2001 and 2002 respectively. Adult lizards (29 females and 15 males) were captured in 2001–2002 and housed in terraria (40 x 60 x 30 cm) with a soil substrate and a thermal gradient from 25–50°C. Eggs were placed in plastic cups with 35 g of moist vermiculite (10 g vermiculite: 8 g water) and incubated at 30°C. Hatchlings were placed in a nursery terrarium away from adults. Adults were re-released at their point of capture. Hatchlings were raised for around 49 days in 2001 and 66 days in 2002 before being released in to the wild.

A replicated study in 2010–2011 at the Bronx Zoo Department of Herpetology, USA (13) found that almost all eggs from captive common chuckwallas *Sauromalus ater* hatched successfully following incubation in suspended incubation containers. Three females each produced a clutch (7–9 eggs/clutch), and 23 of 24 eggs (96%) hatched successfully. Incubation periods ranged from 72–74 days following incubation at 31.5°C, to 79–80 days following incubation at 30°C. One egg became mouldy and was removed after 27 days. All 23 hatchlings survived at least 6 weeks following hatching. In 2010–2011, three clutches of seven, eight and nine eggs each were moved to incubation containers, where they were suspended over wet vermiculite. Incubation temperatures were 30, 30–31 and 31.5°C, and humidity was around 100% in all containers.

A controlled study (year not provided) in laboratory conditions in the USA (14) found that captive-reared western fence lizards *Sceloporus occidentalis* bred successfully in captivity, but that more clutches were laid and fewer eggs were infertile when female lizards were housed individually or in pairs compared to in larger groups. Individually-kept females produced more clutches (3.4 clutches laid/female) and fewer infertile eggs (11% infertile eggs/clutch) compared to females kept in groups of four or eight (2.3–2.7 clutches laid/female; 31.4–37.7% infertile eggs/clutch). Female lizards kept in pairs laid similar numbers of clutches (3.1 clutches/female) and infertile eggs (10.6% infertile eggs) to. Clutch sizes, and the proportion of females that laid eggs were similar between different sized groups (see original paper for details). Eggs from wild-caught western fence lizards were hatched and reared in captivity. In total, 96 nine-month-old female lizards were housed either individually or in groups of two/cage, four/cage or eight/cage (24 lizards/treatment) for the breeding season. One male lizard was randomly assigned to each cage for breeding. Eggs were collected from cages within 12 hours of being laid. Females and eggs were monitored until egg-laying ceased and egg fertility was assessed by 'candling'.

A study in 2011–2013 in a captive setting in the UK (15) reported that a pair of Rio Fuerte beaded lizards *Heloderma exasperatum* bred successfully in captivity. One female beaded lizard produced eight eggs in captivity, four of which hatched successfully. From the four eggs that were not successful, one hatchling died during emergence, one died during development, and two were infertile. In 2011, a pair of adult beaded lizards were introduced into an enclosure (3 x 1.5 x 1.5 m) that contained a substrate of sand and sphagnum moss blocks *Sphagnum* spp. and a range of rocks, branches and plants. Temperatures ranged from 35–40°C in a basking area, and 17–20°C in the rest of the enclosure, and 2 L of water was sprayed each day to increase humidity. Eggs were left to incubate in the enclosure and hatchlings were weighed and moved to separate containers (35 x 20 x 16 cm).

A controlled study in 2008–2011 at Los Angeles Zoo, USA (16) found that giant horned lizards *Phrynosoma asio* bred successfully in captivity, and lower incubation temperatures resulted in longer incubation periods and larger hatchlings. Results were not statistically tested. In 2010–2011, a female produced two clutches of eggs, and 9 of 20 (45%) and 15 of 19 eggs (79%) hatched successfully. In 2011, incubating at 26.5°C rather than 28°C resulted in longer incubation periods (26.5°C: 107–112 days; 28°C: 85–92 days) and larger hatchlings (26.5°C: 1.7–2.1 g; 28°C: 1.4–1.6 g). In 2010, eggs were initially incubated at 31°C but eleven began to wither after a few days. In 2008, one female and two male lizards were housed in a 380 litre tank with a substrate of 80% sand and 20% soil. Temperatures were 28°C, with a basking area at 30–37°C, and 70% humidity. In 2010, eggs were moved to a 3 litre plastic container, placed in vermiculite (4:1 ratio with water by weight), and incubated at 31°C. After a few days the temperature was reduced to 26.5°C and more water was added to the vermiculite (2:1 ratio with water). In 2011, six eggs were incubated at 28°C, and 13 at 26.5°C.

A study in 2012–2014 at The Phoenix Zoo, USA (17) found that Yuman fringe-toed lizards *Uma rufopunctata* bred successfully in captivity. In 2013, five clutches of fertile eggs were produced by three female lizards, with an average clutch size of three eggs. Of 14 viable eggs, 13 were incubated and 10 hatched successfully. One egg was damaged during excavation and one clutch of four eggs produced only a single hatchling that died immediately after emergence. Incubation period ranged from 58 days at 31°C to 74 days at 27.5°C. Ten hatchlings survived for at least six months to a year. All females selected nest boxes with an 8:1 sand to water mixture for laying eggs rather than a 16:1 mixture. Four female and three male lizards were acquired in 2012 and maintained in glass tanks (91 x 46 x 43 cm) with a sand substrate and rocks. Temperatures ranged from 43°C under basking lamps to 28°C. Breeding enclosures contained two nest boxes, one containing an 8:1 and the other a 16:1 sand to water mixture.

A replicated study in 2009–2014 at Perth Zoo, Australia (18) found that banded knob-tailed geckos *Nephrurus wheeleri cinctus* bred successfully in captivity and some offspring survived for at least a year. Four females produced a total of 40 clutches of eggs over four breeding seasons (2–6 clutches/female/year). The total number of eggs and hatching success was not provided, but authors reported that only one egg failed to hatch. In 2009, eleven geckos were acquired (5 females, 6 males) and housed in four enclosures (88 x 55 x 60 cm), each with a sand substrate, a nesting box and various rocks and branches. Ambient temperatures were 19–26°C with a 31–35°C basking area in summer, and 15–22°C and 24–28°C basking area in winter. Eggs were removed and incubated in perlite or vermiculite (1:1 or 2:1 mix with water) at 29–30°C.

A replicated study in 2012–2015 at London Zoo, UK (19) reported that tree-runner lizards *Plica plica* bred successfully in captivity, and one of the captive-bred offspring also went on to bread successfully. One female lizard produced six clutches of eggs (2 clutches/year) over three years and a total of 18 eggs, 11 of which hatched successfully (61%). One of the female captive-bred offspring went on to breed, producing one clutch of two eggs (hatching data not provided). In 2012, one female and two male lizards were acquired and housed in a number of different enclosures, with temperatures ranging from 18–30°C and 33–38.6°C in basking areas. Eggs were removed and placed in plastic containers, partially buried in water-soaked vermiculite, and incubated at 26°C. Hatchlings were placed in a range of difference enclosure types (see paper for details).

A study in 2012–2014 in an outdoor enclosure in San Salvador, Bahamas (20) found that housing wild adult San Salvador rock iguanas *Cyclura rileyi rileyi* together in captivity did result in breeding and egg laying, but only one egg hatched successfully. Six months after bringing wild iguanas into captivity, one young iguana hatched successfully. In the subsequent two years, although gravid females dug burrows and deposited eggs, no hatchlings emerged. A breeding facility was established in May 2012, and eight adult iguanas (3 males and 5 females) were brought into captivity from the wild and housed in an outdoor enclosure (9 x 6 m).

A replicated study in 1991–2015 in seven zoos in Jamaica and the USA (21) found that Jamaican iguanas *Cyclura collei* bred in captivity, though most females laid infertile eggs each year. After 24 years of a captive breeding and head-start programme in one Jamaican zoo, five Jamaican iguanas hatched successfully, of which three were released into the wild and two died prior to release. No breeding took place in zoo exhibit cages. After 19–21 years of a captive-breeding programme in American zoos, 73 iguanas hatched successfully. The first hatchling emerged after 6–8 years, but died. Twenty-four hatchlings emerged after 10–12 years and 48 hatchlings emerged after 16–20 years. The authors reported that Jamaican iguanas were less likely to breed in captivity than other captive *Cyclura* spp. and that in the USA zoos, almost all female iguanas laid infertile eggs annually. In total, 617 Jamaican iguanas were transferred to one zoo in Jamaica (593 individuals in 1991–2015) and six zoos in USA (24 individuals in 1994–2009, see original paper for details) as part of a head-starting and captive-breeding programme.

A replicated study in 2009–2016 in two captive-breeding facilities on Christmas Island and at Taronga Zoo, Sydney, Australia (22) found that captive populations of Lister's geckos *Lepidodactylus listeri* and Christmas Island blue-tailed skink *Cryptoblepharus egeriae* increased in size at one captive-breeding facility and remained relatively stable at the other. Results were not statistically tested. On Christmas Island, captive populations of Lister's geckos grew from 50 in 2012 to 500 in 2016, and captive populations of blue-tailed skinks grew from 150 in 2012 to 750 in 2016. At Taronga zoo, populations of Lister's geckos (70 in 2011 and 70 in 2016) and blue-tailed skinks (100 in 2011 and 220 in 2016) remained relatively stable. In 2009, all Lister's geckos and blue-tailed skinks that could be found on Christmas Island were brought into captivity. From these wild-caught individuals and their offspring, 56 geckos and 83 skinks were transported to Taronga Zoo, and the remaining 70 geckos and 109 skinks were maintained at facilities on Christmas Island. Captive management aimed to maximise retention of genetic diversity (see paper for more details).

A replicated study in 2016–2017 in New South Wales, Australia (23) found that Australian painted dragons *Ctenophorus pictus* bred successfully in captivity, and that hatching success was not affected by incubation temperature but was higher during the early breeding season. Females produced 1–4 clutches, with an average of 4 eggs/clutch. Overall hatching success was 60% (66 of 110 eggs) and hatching success was similar across all incubation temperatures (68% at 28°C; 56% at 30°C; 57% at 32°C). In addition, hatching success was higher for eggs laid earlier in the season (data presented at statistical model results). Wild-caught dragons were housed as breeding pairs in cages (50 x 40 x 35 cm) with a sand substrate, basking area, and sandy area for egg laying. Temperatures fluctuated between 15–25°C. Eggs were removed (110 eggs from 19 females) and placed in individual plastic cups (125 ml) in moist vermiculite (1:5 ratio with water by volume), and the cups were sealed with plastic cling wrap and a rubber band. Eggs from each clutch were split evenly between three incubation temperatures: 28, 30 or 32°C (110 eggs overall). Eggs were checked daily and those that failed during incubation were removed.

(1) Peterson K.H. (1982) Reproduction in captive *Heloderma suspectum*. *Herpetological Review*, 13, 122–124.

(2) Honegger R.E. (1985) Additional notes on the breeding and captive management of prehensile-tailed skink (*Corucia zebrata*). *Herpetological Review*, 16, 21–23.

(3) Sylber C.K. (1985) Eggs and hatchlings of the yellow giant chuckwalla and the black giant chuckwalla in captivity. *Herpetological Review*, 16, 18–21.

(4) Chippindale P. (1991) Captive breeding of the Timor Monitor (*Varanus timorensis similis*). *Herpetological Review*, 22, 52–53.

(5) Ferguson G.W. (1991) Ad-libitum feeding rates, growth and survival of captive-hatched chameleons (*Chamaeleo pardalis*) from Nose Be Island, Madagascar. *Herpetological Review*, 22, 124–125.

(6) Card W. (1994) Double clutching Gould's monitors (*Varanus gouldii*) and Gray's monitors (*Varanus olivaceus*) at the Dallas Zoo. *Herpetological Review*, 25, 111.

(7) Leptien R. & Böhme W. (1994) First captive breeding of *Lacerta* (*Omanosaura*) *cyanura* Arnold, 1972, with comments on systematic implications posed by the reproductive pattern and the juvenile dress. *Herpetozoa*, 7, 3–9.

(8) Gonzalez-Ruiz A., Godinez-Cano E. & Rojas-Gonzalez. I. (1996) Captive reproduction of the Mexican Acaltetepon, *Heloderma horridum*. *Herpetological Review*, 27, 192.
(9) McCoid M.J., Henke S.E. & Hensley R.A. (2005) Husbandry and captive reproduction in *Carlia ailanpalai* (Scinidae). *Herpetological Review*, 36, 292–293.
(10) Mattioli F., Gili C. & Andreone F. (2006) Economics of captive breeding applied to the conservation of selected amphibian and reptile species from Madagascar. *Natura–Società italiana di Scienze naturali e Museo civico di Storia Naturale di Milano*, 95, 67–80.
(11) Regalado R. (2006) Reproduction and growth of seven species of dwarf geckos, *Sphaerodactylus* (Gekkonidae), in captivity. *Herpetological Review*, 37, 13–20.
(12) Santos T., Pérez-Tris J., Carbonell R., Tellería J.L. & Díaz J.A. (2009) Monitoring the performance of wild-born and introduced lizards in a fragmented landscape: implications for ex situ conservation programmes. *Biological Conservation*, 142, 2923–2930.
(13) Baumer M., Foster C.D., Casey B. & Titus V. (2012) Successful incubation of common chuckwalla (*Sauromalus ater*) eggs at the Bronx Zoo using suspended incubation method. *Herpetological Review*, 43, 597–599.
(14) Talent L.G. & Talent S.G. (2013) Effects of crowding on reproductive traits of Western Fence Lizards, *Sceloporus occidentalis*. *Herpetological Conservation and Biology*, 8, 251–257.
(15) Radovanovic A. (2014) Captive husbandry and management of the Rio Fuerte beaded lizard *Heloderma exasperatum*. *The Herpetological Bulletin*, 130, 6–8.
(16) Recchio I., Robertson-Billet M., Rodriguez C. & Haigwood J. (2014) Captive husbandry and reproduction of *Phrynosoma asio* (Squamata: Phrynosomatidae) at the Los Angeles Zoo and Botanical Gardens. *Herpetological Review*, 45, 450–454.
(17) Foster C.D., Tietgen M. & Baumer M. (2015) Yuman Fringe-toed Lizard (*Uma rufopunctata*) care and breeding at the Phoenix Zoo. *Herpetological Review*, 46, 46–49.
(18) McGill, B. (2015) Captive husbandry and breeding of the banded knob-tailed gecko (*Nephrurus wheeleri cinctus*) at Perth Zoo. The Herpetological Bulletin, 134, 6–9.
(19) Harding L., Tapley B., Gill I., Kane D., Servini F., Januszczak I.S., Capon-Doyle J.S. & Michaels C.J. (2016) Captive husbandry and breeding of the tree-runner lizard (*Plica plica*) at ZSL London Zoo. *The Herpetological Bulletin*, 138, 1–5.
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(22) Andrew P., Cogger H., Driscoll D., Flakus S., Harlow P., Maple D., Misso M., Pink C., Retallick K., Rose K., Tiernan B., West J. & Woinarski J.C.Z. (2018) Somewhat saved: a captive breeding programme for two endemic Christmas Island lizard species, now extinct in the wild. *Oryx*, 52, 171–174.
(23) Hansson A. & Olsson M. (2018) The influence of incubation temperature on phenotype of Australian Painted Dragons (*Ctenophorus pictus*). *Herpetologica*, 74, 146–151.

Crocodilians

- **Six studies** evaluated the effects of breeding crocodilians in captivity. Two studies were in the USA[1,5], one was in each of Venezuela[2], Brazil[3] and China[6] and one was a global review[4].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (6 STUDIES)

- **Abundance (1 studies):** One study in China[6] reported that a captive population of Chinese alligators increased from 10,000 to 15,000 individuals over a 10-year period.
- **Reproductive success (4 studies):** Four studies in the USA[1,5], Venezuela[2] and Brazil[3] reported that 1–4 captive females crocodilians, including four captive-born broad-

snouted caiman[3], produced clutches of 17–49 eggs, with hatching successes of 35–86%[1,2,3] or 6%[5].

- **Survival (1 studies):** One study in Brazil[3] reported that 4% of broad-snouted caiman hatchlings died within one week
- **Condition (1 studies):** One global review[4] reported on one study on Chinese alligators that found that captive breeding had a positive effect on genetic variation compared to wild populations.

BEHAVIOUR (0 STUDIES)

A study in 1966–1979 in Ohio, USA (1) reported that Schneider's smooth-fronted caimans *Paleosuchus trigonatus* bred successfully in captivity after 13 years. In 1979, a female produced a clutch of 17 eggs, six of which hatched successfully after 115–118 days of incubation. One hatchling died after two weeks, and the other five survived at least three months. In 1975, a clutch was laid but the eggs rolled into the water and were broken. In 1966–1967, a juvenile pair of caimans were acquired and housed together in a circular pool with a concrete island along with several other crocodilians. After the clutch was lost in 1975, the pair were moved to a new enclosure with dense vegetation and a pool of water. Water temperatures ranged from 21–27°C and air temperatures were 21°C in winter and followed ambient temperatures in summer. Eggs were moved to a Styrofoam container, covered with peat moss and incubated at 29–31°C and 92–100% humidity. After 112 days of incubation, eggs were uncovered, moved gently and a grunting call was made to simulate actions by an adult.

A study in 1987–1989 in Barinas state, Venezuela (2) reported that one of two female Orinoco crocodiles *Crocodylus intermedius* bred successfully in captivity. In 1987–1988, two females produced four clutches of 37–49 eggs each. Hatching success of eggs from one female was 72%, but no eggs produced by the other female hatched. Incubation periods were 78–85 days. In 1989, a further clutch of 52 eggs was laid, but no data on hatching success was available. Yearly survival of hatchlings was 4–50%. In 1987, one male and two female crocodiles were acquired and housed in an outdoor facility with two ponds (20 x 10 x 1 m and 10 x 4 x 1 m) and two sand beaches for egg laying. Nests were incubated under natural conditions.

A study in 1996 in a captive facility in São Paulo, Brazil (3) reported that second-generation captive-bred broad-snouted caiman *Caiman latirostris* bred successfully in captivity. Four female broad-snouted caiman first laid a single clutch each at approximately 10 years old (36–44 eggs/clutch). Hatching success was 40–86% per clutch (81 of 121 eggs hatched). Three hatchlings died within the first week of emerging. Four female and one male broad-snouted caiman were born in captivity in 1986, and maintained in enclosed pens (see original paper for details). Eggs were artificially incubated (see original paper for details).

A review of studies investigating the genetics of captive breeding programmes (4) found that captive breeding reptiles had mixed genetic outcomes in comparison to wild populations. Nine percent of 131 studies related to reptiles. One study on American alligators *Alligator mississippiensis* found that captive breeding had a positive effect on two measures of genetic diversity (measured as expected heterozygosity and number of alleles), but a negative effect on the chance of inbreeding compared to wild populations. Two databases (Web of Science and Zoological Record) were searched for studies investigated the genetics of captive populations up until 2010.

A study in 2012–2015 at the Smithsonian's National Zoological Park, Washington DC, USA (5) found that Cuban crocodiles *Crocodylus rhombifer* bred in captivity, but hatching success of eggs was low. Twenty-six eggs were produced in 2012 and 24 in 2015. A total of three eggs hatched successfully and a further four produced hatchlings after eggs were opened manually or hatchlings were assisted during emergence. All eggs came from a single breeding pair of adult crocodiles, and 10 from each clutch that showed signs of development were incubated (eight in a 1:1 mixture of vermiculite and water, and two in suspended incubation containers). Eggs in suspended incubation containers were suspended over saturated vermiculite or a saturated sponge and standing water. Incubation containers were vented throughout the process and water was added to the vermiculite weekly.

One study in 1982–1985 and 2006–2016 in a captive facility in Xuancheng, Anhui Province, China (6) reported that a captive population of Chinese alligators *Alligator sinensis* increased over a 10-year period. The captive population grew from 10,000 individuals in 2006 to 15,000 in 2016. In 1982–1985, wild alligators (212 individuals) and nests (778 eggs) were brought in to captivity as part of a breeding programme.

(1) Jardine D.R. (1981) First successful propagation of Schneider's smooth-fronted caiman, *Paleosuchus trigonatus*. *Herpetological Review*, 12, 58–60.

(2) Ramo C., Busto B. & Utrera A. (1992) Breeding and rearing the Orinoco crocodile *Crocodylus intermedius* in Venezuela. *Biological Conservation*, 60, 101–108.

(3) Verdade L.M. & Sarkis F. (1998) Age at first reproduction in captive *Caiman latirostris* (Broad-snouted caiman). *Herpetological Review*, 29, 227.

(4) Witzenberger K.A. & Hochkirch A. (2011) Ex situ conservation genetics: a review of molecular studies on the genetic consequences of captive breeding programmes for endangered animal species. *Biodiversity and Conservation*, 20, 1843–1861.

(5) Augustine L. (2016) *Crocodylus rhombifer* (Cuban crocodile). Suspension incubation. *Herpetological Review*, 47, 240–241.

(6) Manolis C., Shirley M., Siroski P., Martelli P., Tellez M., Meurer A. & Merchant M. (2016) *CSG Visit to China, August 2016*. 13pp. IUCN-SSC Crocodile Specialist Group.

Tuatara

- **One study** evaluated the effects of breeding tuatara in captivity. This study was in New Zealand[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Reproductive success (1 study):** One replicated study in New Zealand[1] reported that hatching success of eggs laid in captivity by tuatara was around 50%. The study[1] also found that the first clutches were laid 2–8 years after tuatara were brought into captivity.

BEHAVIOUR (0 STUDIES)

A replicated study in 1990–2007 in artificial enclosures in North Island, New Zealand (1) reported that wild tuatara *Sphenodon punctatus* bred multiple times in captivity but that fewer than half of eggs hatched successfully. Over 16 years, 241 of 553 eggs (44%) laid by tuatara in captivity hatched successfully. Clutches were laid in 13 of 16 years by 15 of 22 females. The first clutches were laid 2–8 years after tuatara were brought into captivity. Hatching success and adult survival varied between tuatara taken from different islands (see original paper for details). Three captive-born females also produced three clutches during the study. In 1990–1992, four populations of tuatara were brought into captivity from four islands (6–15 individuals/island) to one of three captive facilities pending eradication of Pacific rats *Rattus exulans*. Tuatara were housed in predator-proof outdoor enclosures. In 1992–2007, eggs were moved to a separate facility for artificial incubation in dampened vermiculite (see original paper for details). Overall, four clutches were induced and 27 clutches were laid naturally. Hatchlings were returned to their source facility at one week–11 months old. Eggs that perished shortly after being laid (5–16 eggs in two clutches) and eggs laid by artificially-incubated females were excluded from results.

(1) Keall S.N., Nelson N.J. & Daugherty C.H. (2010) Securing the future of threatened tuatara populations with artificial incubation. *Herpetological Conservation and Biology*, 5, 555–562.

14.12. Use artificial insemination

- **One study** evaluated the effects of using artificial insemination on reptile populations. This study was in New Zealand[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Reproductive success (1 study):** One replicated study in New Zealand[1] found that none of 10 artificially inseminated McCann's skinks gave birth within a year of insemination, though around five were gravid after nine months.

BEHAVIOUR (0 STUDIES)

Background

During programmes to rear endangered animals in captivity, in preparation for reintroductions into the wild, artificial insemination may be used to initiate pregnancies. The technique may be used instead of natural mating in situations where animals are being kept at different facilities; to increase pregnancy rates; or where natural mating has failed. It may also be carried out using preserved sperm for purposes of maintaining genetic diversity.

Studies included here are those identified by our searches of conservation journals. It is likely that other relevant studies exist in biological journals that specialize in reproduction.

A replicated study in 2008–2009 in outdoor enclosures in Dunedin, New Zealand (1) reported that artificially inseminated female McCann's skinks

Oligosoma maccanni did not give birth, although approximately half were gravid nine months after insemination. None of 10 artificially inseminated female McCann's skinks gave birth in the year after insemination took place. Two months after insemination, eight of 10 artificially inseminated females were confirmed as ovulating and the authors reported that nine months after insemination approximately five of 10 of the females appeared to be gravid. Ten female McCann's skinks kept in captivity were inseminated in March 2008 using sperm pooled from six males collected over two days (each female received at least 1 x 10^6 motile sperm, see original paper for details). Females were checked for ovulation by palpating the abdominal cavity.

1) Molinia F.C., Bell T., Norbury G., Cree A. & Gleeson D.M. (2010) Assisted breeding of skinks or how to teach a lizard old tricks! *Herpetological Conservation and Biology*, 5, 311–319.

14.13. Freeze sperm or eggs for future use

- We found no studies that evaluated the effects of freezing sperm or eggs for future use on reptile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Captive breeding may result in loss of genetic variation, such that animals that were bred for release back into the wild have reduced fitness. Freezing, or 'cryopreservation', of sperm and eggs, allows them to be stored until they are needed. Gene banks can therefore be created for reptiles, ensuring that species' genetic variation is preserved. It also means that the number of a particular species needed in captivity can be reduced and genes can be swapped between captive facilities. Fewer animals in captivity means that fewer retiles need to be taken from the wild. Freezing can damage cells and so a cryoprotectant, such as dimethyl sulphoxide or glycerol is usually required to protect the cells.

14.14. Alter incubation temperatures to achieve optimal/desired sex ratio

Background

Incubation temperatures (for example warmer or cooler, constant or fluctuating) can influence the sex, size, shape, colour, behaviour, movement ability and post-hatching growth of reptile hatchlings and newborns (Booth *et al*. 2006). Practitioners carrying out conservation activities aimed at maximising hatching success, such as relocating eggs for artificial incubation or to on-beach hatcheries, will therefore need to consider the potential impact of temperature during incubation on hatchlings and populations. Human-induced climate change may also influence the sex ratios of some species of reptiles and limit the viability of populations over time. It may be possible to counter the impacts of climate change on affected populations by managing temperatures during incubation to create appropriate sex ratios.

This action includes studies that test the impact of different temperatures on the sex ratio of reptile hatchlings or newborns. For studies that discuss the effectiveness of relocating nests/eggs for incubation more generally, see *Relocate nests/eggs for artificial incubation*, *Relocate nests/eggs to a nearby natural setting (not including hatcheries)*, and *Relocate nests/eggs to a hatchery*. For studies that discuss the effectiveness of captive breeding more generally, see *Breed reptiles in captivity*.

Due to the number of studies found, this action has been split by species group, though no studies were found for amphisbaenians.

Booth D.T. (2006) Influence of incubation temperature on hatchling phenotype in reptiles. P*hysiological and Biochemical Zoology*, 79, 274–281.

Sea turtles

- **One study** evaluated the effects of altering incubation temperatures to achieve optimal/desired sex ratios on sea turtles. This study was in Canada[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Reproductive success (1 study):** One replicated study in Canada[1] reported that hatching success of two clutches of artificially incubated green turtle eggs was 8% and 62%.

BEHAVIOUR (0 STUDIES)

OTHER (1 STUDY):

- **Offspring sex ratio (1 study):** One replicated study in Canada[1] found that incubating green turtle eggs at higher temperatures resulted in more females hatchlings.

A replicated study in 1995 in an artificial setting in Toronto, Canada (1) found that more female green turtle *Chelonia mydas* hatchlings were produced at higher incubation temperatures compared to at lower temperatures. Hatching success for two clutches of eggs was 8% (7 of 90 eggs) and 62% (67 of 108 eggs). Warmer incubation temperatures produced a higher proportion of female hatchings (30.6°C: 100%; 30.0°C: 50%; 29.4°C: 47%; 28.7°C: 36%; 28.4°C: 18%; 28.2°C: 8%; 27.6°C: 0%). The pivotal temperature for determining sex of hatchlings was estimated at 29.4–30°C. In 1995, green turtle eggs were collected from two nests (90 from one nest and 108 from a second) and brought into an artificial setting and placed in an individual container on a sponge with damp vermiculite. Eggs were incubated at one of seven temperatures between 27.6°C to 30.6°C (14–48 eggs/temperature). Hatching success was assessed, and sex of hatchlings was determined by examining the gonads under a microscope.

(1) Godfrey M.H. & Mrosovsky N. (2006) Pivotal temperature for green sea turtles, *Chelonia mydas*, nesting in Suriname. *The Herpetological Journal*, 16, 55–61.

Tortoises, terrapins, side-necked & softshell turtles

- **Eight studies** evaluated the effects of altering incubation temperatures to achieve optimal/desired sex ratios on tortoise, terrapin, side-necked and softshell turtle populations. Four studies were in the USA[1,2,4,5], two were in Columbia[3,6] and one was in each of Brazil[7] and the Galápagos[8]

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (5 STUDIES)

- **Reproductive success (5 studies):** Four studies (including two replicated studies) in the USA[2,5], Colombia[3] and the Galápagos[8] found that hatching success of alligator snapping turtle[2], Magdalena river turtle[3], western pond turtle[5] and Española giant tortoise[8] eggs varied across the range of temperatures tested. One controlled study in Brazil[7] found that Amazon River turtle nests covered with black plastic sheeting had lower hatching success than uncovered nests.

BEHAVIOUR (0 STUDIES)

OTHER (8 STUDIES):

- **Offspring sex ratio (8 studies):** Seven studies (including three replicated, randomized studies) in the USA[1,2,4,5], Colombia[3,6] and the Galápagos[8] found that hatchling sex ratio of turtles and tortoises was affected by incubation temperature, and that warmer temperatures resulted in more female hatchlings. One controlled study in Brazil[7] found that Amazon River turtle nests covered with black plastic sheeting produced more female hatchlings than uncovered nests.

A replicated, randomized study in 1992 in a laboratory in the USA (1) found that altering the incubation temperature of gopher tortoise *Gopherus polyphemus* eggs resulted in different sex ratios of hatchlings. Cooler incubation temperatures produced more males, whereas warmer temperatures produced more females (26°C: 4 male, 0 female; 29°C: 3 male, 1 female; 32°C: 0 male, 4 female). Overall hatching success was 77% (20 of 26 eggs), and incubation period was longer at lower temperatures (26°C: 115 days; 29°C: 97 days; 32°C: 86 days). In June 1992, four wild-caught tortoises were induced with oxytocin and the 26 eggs produced were randomly assigned to the 26, 29 or 32°C treatment. Eggs were placed in containers of moist vermiculite, and every two weeks containers were rotated, and water levels topped up. After 10 months in captivity 17 hatchlings had died and 13 of these were sexed successfully.

A replicated study in 2002–2004 in a laboratory in Oklahoma, USA (2) found that incubating alligator snapping turtle *Macrochelys temminckii* eggs at higher temperatures resulted in strongly female-biased sex ratios, though the coldest and warmest temperatures resulted in very low hatching success. Warmer incubation temperatures produced almost all female hatchlings (28.5°C or 30.5°C resulting in 97% and 100% of female hatchlings), whereas incubating at 26.5°C resulted in 81% male hatchlings. The coldest (23.0 and 24.5°C) and warmest (31.0°C) incubation temperatures resulted in 0% hatching success, and hatching success was higher at 26.5°C (85%) compared to cooler (26.0°C: 33%) or warmer (30.5°C: 40%) temperatures. Eggs were obtained in 2002 (3 clutches of 15–37 eggs, 88 total) and 2004 (6 clutches of 17–42 eggs, 186 in total) and split evenly between six incubation temperatures in 2002 (23.0, 24.5, 26.0, 26.5, 28.5 and 31.0°C; 12–13 eggs/temperature) and three temperatures in 2004 (53 eggs at 26.5°C; 51 at 28.5°C; 47 eggs at 30.5°C). Eggs were incubated in damp vermiculite (1:1 ratio with water by mass). Hatching success was assessed and hatchlings were sexed 267–278 after hatching by observing gonads via a non-lethal surgical procedure.

A study in 2005–2007 in laboratory conditions in Colombia (3) found that higher incubation temperatures increased Magdalena river turtle *Podocnemis lewyana* hatching success, and that females were only produced above a temperature threshold. In the first year, eggs incubated at 33.0°C had a higher hatching success rate (28 of 29, 97% eggs hatched) than eggs incubated at 28.0°C (14 of 38, 37% eggs hatched) and all hatchlings at both temperatures were male. In the second year, five of 10 eggs (50%) incubated at 33.4°C and seven of seven eggs (100%) incubated at 34.7°C produced female hatchlings. The authors reported that nests monitored in the field showed a similar pattern, with the coldest nests having lower hatching rates (see original paper for details). In 2006, river turtle eggs were obtained from 28 nests and incubated in a laboratory at: 28.0°C (38 eggs), 29.5°C (43 eggs), 32.0°C (39 eggs) and 33.0°C (29 eggs). In 2007, river turtle eggs were obtained by inducing four female river turtles using an injection of oxytocin and incubated at: 33.4°C (14 eggs) and 34.7°C (13 eggs). Eggs were monitored through to hatching. Only hatching success data from 2006 laboratory eggs are included as the authors reported that the use of oxytocin to obtain eggs in 2007 may have affected hatching rates. Natural nests were monitored in the field in 2005 (8 nests) and 2006 (11 nests) for incubation temperature and hatching success.

A before-and-after study in 2006–2009 at a captive breeding facility in southern California, USA (4) found that radiated tortoise *Astrochelys radiata* most eggs incubated at 28.9°C or higher produced female hatchlings. Results were not statistically tested. At 28.9°C, twenty-three of 25 hatchlings (92%) were female, and at 30°C, all 29 hatchlings were female. Eggs from captive tortoises were collected and incubated in modified wine coolers at 28.9°C in 2006–2007, and at 30°C in 2008–2009. Eggs were incubated in vermiculite and water at a 2:1 ratio.

A replicated, randomized study in 2008–2009 in laboratory conditions in California, USA (5) found that altering incubation temperatures of western pond turtle *Actinemys marmorata* eggs resulted in variable hatching success, and that no female hatchlings were produced at lower temperatures. Hatching success varied with temperature, with the highest reported success rate at 29°C (26 of 28, 93%) and the lowest at 31°C (3 of 7, 43%). Hatching success at other temperatures (26, 27, 28 or 30°C) ranged from 68–82%. Eggs at 26–27°C produced all male hatchlings and those at 28–29°C were highly skewed towards males (28°C: 92% males; 29°C: 85% males), while those at 30°C produced all females. In 2008–2009, eggs were obtained from 44 wild turtle nests. Eggs were distributed evenly between five plastic containers that were partially filled with moist vermiculite (5:1 ratio with water by volume) and incubated at constant temperatures. Five temperatures were chosen in 2008, and these were all decreased by 1°C in 2009, resulting in the following number of eggs/treatment: 15 eggs at 26°C; 28 at each of 27, 28 and 29°C; 25 at 30°C; seven at 31°C. In 2009, the sex of fifty-nine turtles was determined through a non-lethal surgical procedure that allowed gonads to be observed (see paper for details).

A replicated, randomized study in 2012 in laboratory conditions in Columbia (6) found that incubating Magdalena River turtle *Podocnemis lewyana* eggs at lower temperatures produced more male hatchlings, and higher temperatures produced more females. Hatching success ranged from 57–100%. Lower incubation temperatures resulted in fewer female hatchlings (29°C: 8% female; 31°C: 18%; 34.7°C: 86%). When incubating at 29°C, a 10-day high temperature pulse resulted in more female hatchlings compared to the constant temperature if it came during day 21–30 (37% female), but a similar number if it came at day 31–50 (4–20% female). At 31°C, a pulse during day 21–50 resulted in more females than the constant temperature (32–67% female). In 2012, a total of 227 eggs were collected from 10 nests (14 beaches searched). Eggs were incubated at either a constant temperature (29, 31 or 34.7°C; 30–31 eggs/temperature), or at 29 or 31°C with a 10-day period at a high temperature (35.1–35.5°C) during day 21–30, 31–40 or 41–50 (19–25 eggs/treatment). Sex of 20 individuals was determined by assessing gonadal histology. Detailed morphology of these 20 individuals was used to estimate the sex of all hatchlings (see paper for details).

A controlled study in 2003 on a sandy beach in Amazonas, Brazil (7) found that covering six-tubercled Amazon River turtle *Podocnemis sextuberculata* nests with black plastic sheeting increased the proportion of female hatchlings, but decreased hatching success. River turtle nests covered with black plastic sheeting produced half the number of male hatchlings (1.5 of 5, 30% hatchlings/nest were male) compared to uncovered nests (3.3 of 5, 66% hatchlings/nest were male) and covered nests had lower hatching success (80%) than uncovered nests (92%). In September-November 2003, thirty turtle nests laid on a river-side beach (2 km long, 600 m wide) in a reserve were monitored from within 12 hours of being laid through to hatchling emergence. Fifteen of 30 nests were covered with a sheet of black plastic (0.1 mm thick covering a 2 m^2 area) in order to influence hatchling sex ratios. The remaining fifteen nests were monitored but not covered. Black plastic was removed after 50 days of incubation and nests were covered with nets to capture hatchlings as they emerged. Sex ratios were determined by sacrificing five hatchlings/nest and carrying out an examination.

A study in 1986 in a captive rearing facility in Galápagos, Ecuador (8) found that less than half of artificially incubated Española giant tortoise *Chelonoidis hoodensis* eggs hatched in captivity and that the sex ratio was temperature dependent. Results were not statistically tested. Española giant tortoise eggs artificially incubated at 25.5°C had a hatching success of 16% and 10 of 11 (91%) sexed hatchlings were male. Eggs incubated at 29.5°C had a hatching success of 40% and five of 15 (33%) sexed hatchlings were male. No eggs artificially incubated at 33.5°C hatched successfully. In 1986, giant tortoise eggs laid in captivity as part of a head-starting programme were incubated at three different temperatures: 25.5, 29.5 and 33.5°C (67 eggs/temperature, 189 total eggs). Eggs were placed in plastic boxes with damp vermiculite, covered and put in incubation chambers. Hatchlings were sexed by direct observation, examination of dead hatchlings' gonads (35 individuals) or key-hole surgery (15 individuals). Data from six hatchlings that hatched earlier in the season in the same facility were included in the results.

(1) Burke R.L., Ewert M.A., McLemore J.B. & Jackson D. R. (1996) Temperature-dependent sex determination and hatching success in the gopher tortoise (*Gopherus polyphemus*). *Chelonian Conservation and Biology*, 2, 86–88.
(2) Ligon D.B. & Lovern M.B. (2009) Temperature effects during early life stages of the alligator snapping turtle (*Macrochelys temminckii*). *Chelonian Conservation and Biology*, 8, 74–83.
(3) Paez V.P., Correa J.C., Cano A.M. & Bock B.C. (2009) A comparison of maternal and temperature effects on sex, size, and growth of hatchlings of the Magdalena River Turtle (*Podocnemis lewyana*) incubated under field and controlled laboratory conditions. *Copeia*, 2009, 698–704.
(4) Kuchling G., Goode E. & Praschag P. (2013) Endoscopic imaging of gonads, sex ration, and temperature-dependent sex determination in juvenile captive-bred radiated tortoises, *Astrochelys radiata*. *Chelonian Research Monographs*, 6, 113–118.
(5) Geist N.R., Dallara Z. & Gordon R. (2015) The role of incubation temperature and clutch effects in development and phenotype of head-started Western pond turtles (*Emys marmorata*). *Herpetological Conservation and Biology*, 10, 489–503.
(6) Gómez-Saldarriaga C., Valenzuela N. & Ceballos C.P. (2016) Effects of incubation temperature on sex determination in the endangered Magdalena river turtle, *Podocnemis lewyana*. *Chelonian Conservation and Biology*, 15, 43–53.
(7) Eisemberg C.C., Drummond G.M. & Vogt R.C. (2017) Boosting female hatchling production in endangered, male-biased turtle populations. *Wildlife Society Bulletin*, 41, 810–815.
(8) Sancho A., Gutzke W.H.N., Snell H.L., Rea S., Wilson M. & Burke R.L. (2017) Temperature sex determination, incubation duration, and hatchling sexual dimorphism in the Española Giant Tortoise (*Chelonoidis hoodensis*) of the Galápagos Islands. *Amphibian & Reptile Conservation*, 11, 44–50.

Snakes & lizards

- **Four studies** evaluated the effects of altering incubation temperatures to achieve optimal/desired sex ratios on snake and lizard populations. Two studies were in each of the USA[1,4] and China[2,3].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (3 STUDIES)

- **Reproductive success (3 studies):** Two replicated studies (including one randomized study) in China[3] and the USA[4] found that toad-headed agama hatching success was lowest at the highest incubation temperature tested[3] and southern alligator lizard hatching success was highest at intermediate temperatures[4].One randomized study in the USA[1] found that survival of garter snake offspring was highest when females were maintained at intermediate temperatures

BEHAVIOUR (0 STUDIES)

OTHER (4 STUDIES):

- **Offspring sex ratio (4 studies):** Three replicated studies (including two randomized studies) in China[2,3] and the USA[4] found that hatchling sex ratio of stripe-tailed ratsnakes[2], toad-headed agamas[3] and southern alligator lizard[4] was not affected by incubation temperature. One randomized study in the USA[1] found that sex ratio of live garter snake offspring was not affected by the temperature females were maintained at.

A randomized study (year not provided) in laboratory conditions in California, USA (1) found that maintaining pregnant female garter snakes *Thamnophis elegans* at intermediate temperatures in captivity increased overall embryo survival rates, male offspring stillbirths was also reduced at higher temperatures and temperature did not affect live hatchling sex ratios. When pregnant female garter snakes were maintained at 26.6°C, embryo survival rate was higher than at lower (21–24°C) or higher temperatures (28–33°C; data reported as model outputs). Rates of male offspring stillbirths reduced at intermediate and higher temperatures (data reported as model outputs). Incubation temperature did not affect the sex ratio of live offspring (see paper for details). Seventy-four wild pregnant female garter snakes were brought into captivity and maintained at one of nine constant temperatures (21, 24, 26, 27, 28, 29, 30, 32, and 33°C; 2–14 females/temperature) until giving birth. Females were on average in the 20th day of pregnancy when temperature management began. In total 504 snakes were born.

A replicated, randomized, study in 1998 in a laboratory in Zhejiang, China (2) found that altering the incubation temperature of stripe-tailed ratsnake *Elaphe taeniura* eggs did not affect the sex ratio of hatchlings. The ratio of males to females varied from 2:5 to 13:6 and was not influenced by temperature (result presented as statistical test). In 1998, thirteen captive-born gravid females were acquired and housed in a wire cage (200 x 80 x 80 cm) at 30°C. Eggs were incubated at 22, 24, 27, 30 or 32°C, with eggs from each clutch split evenly between temperatures. Eggs were incubated individuals in covered plastic jars in vermiculite and water at a ratio of 1:2. Hatchlings were euthanized by freezing to -15°C to allow their sex to be determined.

A replicated study in 2011 in Gansu, China (3) found that altering the incubation temperature of eggs from two species of toad-headed agamas *Phrynocephalus przewalskii* and *Phrynocephalus versicolor* did not influence the sex ratio of hatchlings. Sex ratio of hatchlings for *Phrynocephalus przewalskii* (61:53 ratio of females to males) and *Phrynocephalus versicolor* (50:36 ratio of females to males) were not affected by incubation temperature or moisture content of incubation medium. In addition, the highest temperature resulted in lower hatching success for both species (*Phrynocephalus przewalskii*: 34°C: 32–36%; 26–30°C: 40–53%; *Phrynocephalus versicolor*: 34°C: 11–22%; 26–30°C: 52–76%), although this result was not tested statistically. In 2011, wild female lizards of both species were captured and housed in groups of 15 in cages (800 x 360 x 400 mm) with a sand substrate. Temperatures of 25–37°C were available during the day and were 20°C at night. Eggs were collected (*Phrynocephalus przewalskii*: 263 eggs from 101 females; *Phrynocephalus versicolor*: 185 eggs from 66 females) and assigned to three temperature (26, 30, 34°C) and two moisture level (2 g water/5 g vermiculite, 2 g water/8 g vermiculite) treatments. Eggs were incubated in plastic containers (150 ml).

A replicated, randomized study in 2010–2011 in laboratory conditions in Iowa, USA (4) found that the sex ratio of southern alligator lizard *Elgaria multicarinata* hatchings was not affected by incubation temperature and that hatching success was highest at intermediate temperatures. Sex ratio was not affected by incubation temperature, and overall, 15 of 21 (71%) hatchlings were male. In addition, hatching success was higher at intermediate temperatures (19 of 24, 79% at 26°C; 21 of 24, 88% at 28°C) than at the coolest (2 of 6, 33% at 24°C) or highest temperatures tested (11 of 25, 44% at 30°C; 0 of 6, 0% at 32°C), though this result was not tested statistically. Eggs were incubated in individual glass jars (140 ml), half buried in moist vermiculite (water potential of -150 kPa), and jars were covered with clear plastic wrap. Eggs were split between five temperature treatments: 24°C (6 eggs); 26°C (24 eggs); 28°C (24 eggs); 30°C (25 eggs); and 32°C (6 eggs). Sex was determined by assessing gonadal morphology (at six months) or histology (at 30 days).

(1) O'Donnell R.P. & Arnold S.J. (2005) Evidence for selection on thermoregulation: Effects of temperature on embryo mortality in the garter snake *Thamnophis elegans*. *Copeia*, 2005, 930–934.
(2) Du W.G. & Ji X. (2008) The effects of incubation temperature on hatching success, embryonic use of energy and hatchling morphology in the stripe-tailed ratsnake *Elaphe taeniura*. *Asiatic Herpetological Research*, 11, 24–30.
(3) Tang X., Yue F., Ma M., Wang N., He J. & Chen, Q. (2012) Effects of thermal and hydric conditions on egg incubation and hatchling phenotypes in two *Phrynocephalus* lizards. *Asian Herpetological Research*, 3, 184–191.
(4) Telemeco R.S. (2015) Sex determination in southern alligator lizards (*Elgaria multicarinata*; Anguidae). *Herpetologica*, 71, 8–11.

Crocodilians

- **Three studies** evaluated the effects of altering incubation temperatures to achieve optimal/desired sex ratios on crocodilian populations. Two studies were in Argentina[2,3] and one was in China[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Reproductive success (2 studies):** Two replicated, randomized study in Argentina[2,3] found that hatching success of broad-snouted caiman eggs was similar across all temperatures tested.

BEHAVIOUR (0 STUDIES)

OTHER (3 STUDIES):

- **Offspring sex ratio (3 studies):** Two replicated, randomized studies in Argentina[2,3] found that hatchling sex ratio of broad-snouted caimans was affected by temperature, and that warmer temperatures resulted in fewer females. One replicated study in China[1] found that exposing Chinese alligator eggs to short periods of high temperatures during incubation resulted in fewer female hatchlings.

A replicated study in 1988–1989 in laboratory conditions in Anhui, China (1) found that short exposure to high temperatures during incubation of Chinese alligator *Alligator sinensis* eggs resulted in fewer female hatchlings compared to when temperatures were kept constant. Results were not statistically tested. Hatching success ranged from 90–100% (10–20 eggs/group). Less females were produced from eggs exposed to 34°C for 4–7 days (0 of 10 to 2 of 15, 0–13% female hatchlings) compared to when eggs were incubated at 31–32°C (15 of 20, 75% female hatchlings). Eggs were incubated at 31–32°C, and nine groups of 10 eggs were exposed to 34°C for four continuous days starting on the 14th and 24th day of incubation. One group of 16 eggs was exposed to seven days at 34°C from the 24th–31st day of incubation. An additional group of 20 eggs was incubated at 31–32°C for the whole duration. Tissue samples were assessed to determine the sex of hatchlings.

A replicated, randomized study in 2013 in laboratory conditions in Santa Fe province, Argentina (2) found that altering the incubation temperature of broad-snouted caiman *Caiman latirostris* eggs did not affect hatching success, but that females were only produced below a temperature threshold. Hatching success was similar across all temperatures (26 of 30, 88% at 31°C; 25 of 29, 85% at 33°C; 23 of 29, 78% at 34°C). Incubation at 31°C produced all females (46 eggs), whereas incubation at 33°C (45 eggs) and 34°C (43 eggs) produced all males. In 2013, a total of 134 viable eggs were collected from four wild nests and clutches were split evenly between three incubation temperatures (31, 33 or 34°C) with two groups/temperature. Eggs were incubated in moist vermiculite at high humidity. Forty-six eggs were dissected during development, just after the thermosensitive period when sex is determined. Sex was assessed by histological methods (46 embryos and 14 hatchlings) or by visual examination four months after hatching (74 hatchlings).

A replicated, randomized study (year not provided) in Santa Fe province, Argentina (3) found that when altering incubation temperatures of broad-snouted caiman *Caiman latirostris* eggs, lower temperatures resulted in a higher number of female hatchlings compared to higher temperatures. At 31°C, all hatchlings were female, and at 33°C and 34°C all hatchlings were male (number of eggs/treatment not provided). At 32°C an average of 72% of hatchlings were female, but this varied from 17–100% depending on the nest of origin. Hatching success varied from 78–91% and was not affected by incubation temperature. A total of 172 eggs that were judged to be viable (by presence of opaque banding on egg) were collected from nine wild nests. Eggs were incubated at 32 or 33°C in the first year, and 31, 33 or 34°C in the second year. In both years, there were two groups/temperature, and eggs were split evenly between groups (number/treatment not provided). A total of 141 hatchlings were kept in captivity for four months, after which point sex was determined using histological methods (100 individuals) or by a visual examination (24 individuals).

(1) Zhang Z.D. (1995) Research on the sex sensitive period during the incubation of Chinese alligator eggs. *Asiatic Herpetological Research*, 6, 157–160.

(2) Parachú Marcó M.V.., Piña C.I., Somoza G.M., Jahn G.A., Pietrobon E.O. & Iungman J.L. (2015) Corticosterone plasma levels of embryo and hatchling broad-snouted caimans (*Caiman latirostris*) incubated at different temperatures. *South American Journal of Herpetology*, 10, 50–57.

(3) Parachú Marcó M.V., Leiva P.M.D.L., Iungman J.L., Simoncini M.S. & Piña C.I. (2017) New evidence characterizing temperature-dependent sex determination in broad-snouted caiman, *Caiman latirostris*. *Herpetological Conservation and Biology*, 12, 78–84.

Tuatara

- **Two studies** evaluated the effects of altering incubation temperatures to achieve optimal/desired sex ratios on tuatara populations. Both studies were in New Zealand[1,2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (0 STUDIES)

OTHER (2 STUDIES):

- **Offspring sex ratio (2 studies):** Two replicated studies (including one controlled study) in New Zealand[1,2] found that hatchling sex ratio of tuatara was affected by temperature, and that warmer temperatures resulted in more males.

A replicated study in 1998 in a captive setting in New Zealand (1) found that incubating tuatara *Sphenodon punctatus* eggs at higher temperatures resulted in more male hatchlings compared to cooler temperatures. Results were not statistically tested. More male hatchlings were produced at the highest incubation temperature (22°C: 100% of 113 hatchlings were male) compared to the intermediate temperature (21°C: 4% of 80 hatchlings were male) and lowest temperature (18°C: 0% of 105 hatchlings were male). In 1998, a total of 320 eggs were collected either from natural nests (154 eggs from 29 clutches) or by inducing females to lay eggs with oxytocin (166 eggs from 21 clutches). Eggs were incubated in moist vermiculite in plastic containers, with clutches divided equally for incubation at 18°C, 21°C or 22°C. The sex of young tuatara was determined one year after hatchling using a surgical procedure.

A replicated, controlled study in a captive setting in Wellington, New Zealand (2) found that incubating eggs of two populations of tuatara (*Sphenodon guntheri* and *Sphenodon punctatus*) at higher temperatures produced more male hatchlings. Incubating eggs at higher temperatures resulted in more male offspring (22.1–24°C: 100% of 7–113 eggs produced males) compared to at lower temperatures (18–18.3°C: 0–8% of 12–105 eggs produced males). For one population (*Sphenodon guntheri*), males were produced above 21.6°C, and for the other population (*Sphenodon punctatus*), males were produced above 22.0°C. In 2000, a total of 71 *Sphenodon guntheri* eggs were collected from North Brother Island by inducing gravid females to lay eggs with oxytocin (49 eggs) or removing eggs from nests (22 eggs). Eggs were placed in moist vermiculite and randomly assigned to incubate at 18°C, 21°C, 22°C or 23°C. The sex of these hatchlings was determined via a surgical procedure (see paper for details). In 2003, fifteen eggs from a captive female (*Sphenodon punctatus*) were incubated at 18°C for seven weeks before being moved to 21.5°C (7 eggs) or 24.1°C (8 eggs). For eggs that failed to develop fully, sex could still be determined in some cases. Data from a number of other studies on incubation temperatures and sex ratios from 1989–1991 and 1999 were also included for comparison (see paper for details).

(1) Nelson N.J., Thompson M.B., Pledger S., Keall S.N. & Daugherty C.H. (2004) Egg mass determines hatchling size, and incubation temperature influences post-hatching growth, of tuatara *Sphenodon punctatus*. *Journal of Zoology*, 263, 77–87.

(2) Mitchell N.J., Nelson N.J., Cree A., Pledger S., Keall S.N. & Daugherty C.H. (2006) Support for a rare pattern of temperature-dependent sex determination in archaic reptiles: evidence from two species of tuatara (Sphenodon). *Frontiers in Zoology*, 3, 1–12.

14.15. Maintain wild-caught, gravid females in captivity during gestation

- **Seven studies** evaluated the effects on reptile populations of maintaining wild-caught, gravid females in captivity during gestation. Two studies were in the USA[1,6] and New Zealand[2,3] and one was in each of Japan[4], Iran[5] and Mexico[7].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (7 STUDIES)

- **Reproductive success (7 studies):** Five replicated studies in the USA[1,6], Japan[4], Iran[5] and Mexico[7] found that varying numbers of wild-caught snakes[1,4,5,7] and lizards[6] gave birth to live young[1,5,7] or laid eggs that hatched successfully [4,6] in captivity. One study[6] also found that eggs laid in artificial nest chambers had higher hatching success than those laid outside of the chambers. One study in New Zealand[2] found mixed effects of providing different basking conditions on the number of McCann's skinks and common geckos that gave birth successfully. One controlled study in New Zealand[3] found that McCann's skinks in captivity that were treated for mites completed pregnancy more often and produced more viable offspring compared to skinks not treated.

BEHAVIOUR (0 STUDIES)

Background

Bringing wild, gravid, female reptiles into captivity temporarily until either eggs are laid or, in the case or viviparous reptiles, young are born may improve the hatching or birth rate of young reptiles. This could help slow the decline, maintain or increase population size in populations where females are vulnerable during the breeding season or gestation period.

A replicated study in 1995–2001 in a captive setting in Illinois, USA (1) found that gravid plains gartersnakes *Thamnophis radix* maintained in captivity produced some live offspring. From 38 litters, 473 offspring were alive (79%) and 128 were stillborn (21%). From 18 litters obtained by inducing females with oxytocin, 343 offspring were alive (67%) and 112 stillborn (33%). In 1995–2001, gravid females were captured (number not given) and maintained in captivity in individual glass aquaria (40 l) until giving birth. The room was kept at 24–26°C (32°C at one end of aquarium) and humidity at 50%.

A study in 2005–2008 in rocky grassland and laboratory conditions in South Island, New Zealand (2) found that most wild pregnant McCann's skinks *Oligosoma maccanni* and common geckos *Hoplodactylus maculatus* kept in captivity gave birth, although gestation success in skinks depended on the basking regime that they were exposed to in captivity. Female McCann's skinks exposed to basking temperatures for 3.5 days/week had lower gestation success (53% success) than skinks exposed to basking temperatures for 5 or 7 days/week (78–83% success). Similar numbers of female common geckos developed at least one viable offspring regardless of basking regime (3.5 days/week: 80% success; 5 days/week: 90% success; 7 days/week: 80% success). Most geckos that developed full-term embryos required inducement (see original paper for details). Clutch sizes were similar between different basking regimes for both skinks and geckos (see original paper for details). Pregnant female McCann's skinks and common geckos were collected from the wild in November 2005 and October 2007 and kept in one of three basking regimes, with heat provided for: 8 hours/day for 3.5 days/week (17 skinks, 10 geckos), 5 days/week (18 skinks, 10 geckos) or 7 days/week (23 skinks, 10 geckos). Lizards were monitored until they gave birth and some lizards were induced when over-gestation was apparent using the hormone arginine vasotocin (see original paper for details). Gestation was considered a success when at least one viable offspring was delivered.

A controlled study in 2004 and 2007 in laboratory conditions in South Island, New Zealand (3) found that wild-caught pregnant female McCann's skinks *Oligosoma maccanni* gave birth in captivity, but pregnancy success and offspring viability was improved when skinks were treated for mites. When mites were treated with vegetable oil, the majority of wild-caught pregnant female McCann's skinks gave birth successfully (22 of 30 skinks completed pregnancy successfully, 2 of 30 skinks had partially successful pregnancies), whereas when mites were not treated, most pregnancies were not successful (1 of 17 skinks had a partially successful pregnancy). Female McCann's skinks treated for mites produced more viable offspring (2.6 offspring/female), compared to when mites were not treated (0.1 offspring/female). Two weeks after initial treatment with oil, 14 of 30 female skinks showed signs of mites still being present. After 28 days (and two treatments of oil), no live mites were observed. In October 2004 and 2007, pregnant female McCann's skinks were taken from the wild and maintained in controlled temperature and lighting conditions in individual containers (2004: 17 individuals; 2007: 30 individuals; see original paper for details). In 2004, all skinks had scale mites and were not treated. In 2007, all skinks were treated for mites using sunflower oil following capture. Skinks were checked for mites and retreated with oil as necessary on the 14th day (all skinks oiled), 28th day (only those skinks with raised scales were re-oiled) and 56th day (no skinks were re-oiled) following capture.

A replicated study in 2007 in Japan (4) found that wild-caught, gravid female pit vipers *Ovophis okinavensis* that were maintained in captivity laid eggs that hatched successfully. Authors report that 22 eggs from seven clutches hatched successfully. In 2007, six gravid female snakes were brought into captivity and housed individually in plastic cages. Eggs were temporarily removed for measuring and then returned to the cage to incubate with the female.

A replicated study 2008–2009 in Tehran, Iran (5) found that two of nine wild-caught, gravid Latifi's vipers *Montivipera latifii* gave birth to live young in captivity. In 2008, one female produced a single live young snake, while seven other females produced only stillborn young or infertile egg masses. In 2009, one gravid female produced a litter of 10 live young. In 2008–2009, a total of 26 wild vipers, including nine gravid females, were captured and housed in vivaria of various sizes. Peat moss was provided as a substrate, along with broken flowerpots for cover. Temperatures range from 30–32°C under a heat lamp and 26–28°C elsewhere in the enclosure, and humidity averaged 40–50%.

A replicated study in 2011 in laboratory conditions in Oklahoma, USA (6) found that all wild gravid female eastern collared lizards *Crotaphytus collaris* brought into captivity laid eggs, but that only eggs laid inside artificial nests hatched. All 17 wild-caught gravid female eastern collared lizards laid eggs in captivity (one clutch/individual, 5–9 eggs/clutch). Twelve lizards laid eggs inside artificial nest chambers (74 total eggs) and these eggs had a 62% hatching success (46 of 74 eggs hatched). Five lizards laid eggs outside of artificial nest chambers (29 total eggs) and none of these eggs hatched (23 eggs were desiccated when found after being laid and six eggs became mouldy during incubation). Seventeen gravid female lizards were caught in the Glass Mountains and moved to a laboratory where they were housed individually in partitioned wooden and metal-mesh cages. Each cage section (80 x 40 x 40 cm) contained gravel substrate, artificial lighting and an artificial nest made from bricks and sand/peat moss (see original paper for details). Lizards were fed and watered regularly. Eggs were moved for artificial incubation within 16 hours of being laid and adult lizards were returned to their capture site.

A replicated study in 1991–2004 in laboratory conditions in the State of Mexico, Mexico (7) found that wild-caught, gravid Mexican garter snakes *Thamnophis eques* and blackbelly garter snakes *Thamnophis melanogaster* successfully gave birth to live offspring in captivity. Mexican garters produced 275 live offspring and 13 dead offspring from 21 litters, and blackbelly garters produced 325 live, and 15 dead offspring from 43 litters. The sex ratio for Mexican garters was even (125 males, 146 females, and 4 unsexed), whereas blackbelly garters produced more than twice as many female as male offspring (99 males and 226 females). In 1991–2004, twenty Mexican garter snakes and 43 blackbelly garter snakes that were found to be gravid (by palpating for presence of embryos) were brought into captivity. Snakes were maintained in individual terraria with a paper substrate and a water bowl. Temperatures ranged from 20–25°C. Two to three weeks after birth, adult snakes and their offspring were released where they had been captured.

(1) King R.B. & Stanford K.M. (2006) Headstarting as a management tool: a case study of the plains gartersnake. *Herpetologica*, 62, 282–292

(2) Cree A. & Hare K.M. (2010) Equal thermal opportunity does not result in equal gestation length in a cool-climate skink and gecko. *Herpetological Conservation and Biology*, 5, 271–282.

(3) Hare K.M., Hare J.R. & Cree A. (2010) Parasites, but not palpation, are associated with pregnancy failure in a captive viviparous lizard. *Herpetological Conservation and Biology*, 5, 536–570.

(4) Kadota Y., Kidera N. & Mori A. (2011) One day to hatch: calcium poor eggshells and maternal care in Ovophis okinavensis (Squamata: *Viperidae*). *Herpetological Review*, 42, 26–29.

(5) Kian N., Kaboli M., Karami M., Alizadeh A., Teymurzadeh S., Khalilbeigi N., Murphy J.B. & Nourani E. (2011) Captive management and reproductive biology of Latifi's viper (*Montivipera latifii*) (Squamata: Viperidae) at Razi Institute and Tehran University in Iran. *Herpetological Review*, 42, 535–539.

(6) Santoyo-Brito E., Anderson M.L. & Fox S.F. (2012) An artificial nest chamber for captive *Crotaphytus collaris* that increases clutch success and promotes natural behaviour. *Herpetological Review*, 43, 430–432.

(7) Manjarrez J. & San-Roman-Apolonio E. (2015) Timing of birth and body condition in neonates of two gartersnake species from Central Mexico. *Herpetologica*, 71, 12–18.

14.16. Use hormones and/or other drugs during captive-breeding programmes to induce reproduction/birth

- **Nine studies** evaluated the effects on reptile populations of using hormones and/or other drugs during captive-breeding programmes to induce reproduction/birth. Three studies were in each of the USA[3,5,6] and New Zealand[2,4,7] and one study was in each of the Netherlands[1], China[8] and Japan[9].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (9 STUDIES)

- **Reproductive success (9 studies):** Three of four replicated, controlled studies (including one before-and-after study) in the USA[3,5,6] and China[8] found that plains gartersnakes[3], eastern painted turtles[5] and red-eared sliders[6] induced with oxytocin produced a similar percentage of live young[3] compared to individuals that were not induced and laid eggs with similar hatching success[5] or laid a similar number of eggs[6] compared to what was observed in wild nests. The other study[8] found that 25% of eggs from hormone-injected (luteinizing hormone and gonadotropin) four-eyed turtles were fertile, compared to 7–52% for females that were not injected or injected with a saline solution. One study[5] also found mixed effects of different combinations of hormones and other drugs on inducing 13 turtle species. Five studies (including one before-and-after study) in the Netherlands[1], New Zealand[2,4,7] and Japan[9] found that oxytocin[1,2,4], arginine vasotocin[7] and follicle-stimulating hormone[9] induced egg laying/birth in yellow-headed box turtles[1], tuatara[2,4] and common geckos[7] or ovulation in hawksbill turtles[9]. One study[1] also found that only one yellow-headed box turtle female produced fertile eggs.

BEHAVIOUR (0 STUDIES)

Background

Captive animals do not always breed successfully under artificial conditions. Reproductive technologies such as treatment with hormones or other drugs to induce ovulation or sperm production are techniques that can be used in an attempt to increase breeding success by reptiles in captive facilities. It may be appropriate in some circumstances to use hormone or other treatments on wild individuals, for example when it is not desirable to bring them into captivity. Hormone stimulation protocols are often species specific.

A study in 1989–1992 in a captive setting in the Netherlands (1) found that egg laying could be induced with injections of calcium and oxytocin in yellow-headed box turtles *Cuora aurocapitata*. Two females laid two eggs each within two hours of treatment. One female had laid an egg prior to treatment, and the second female laid an additional egg 12 days after treatment. Two of three eggs from one female hatched successfully, whereas none from the other did. In 1989–1990, two pairs of turtles were acquired and males were introduced to both females for mating purposes. Females were injected with calcium (two doses at 60–80 mg/kg, 1–2 h apart) under the skin in the rear leg, followed by 6 IU/kg of oxytocin intramuscularly one hour later.

A study in 1998 in a captive setting in New Zealand (2) found that female tuatara *Sphenodon punctatus* could be induced to lay eggs using oxytocin. A total of 166 eggs from 21 clutches were produced by inducing females with oxytocin (total number of injected females not provided). In 1998, females were induced to lay eggs with an injection of synthetic oxytocin (Oxytocin-s, 10IU/ml).

A replicated, controlled study in 1995–2001 in a captive setting in Illinois, USA (3) found that inducing gravid plains gartersnakes *Thamnophis radix* with oxytocin resulted in similar birth success compared to females not induced. Results were not statistically tested. From 18 broods obtained by inducing females with oxytocin, 343 offspring were alive (67%) and 112 stillborn (33%), and from 38 litters obtained with no oxytocin, 473 offspring were alive (79%) and 128 were stillborn (21%). In 1995–2001, gravid females were captured (number not given) and maintained in captivity in individual glass aquaria (40 l) until giving birth. The room was kept at 24–26°C (32°C at one end of aquarium) and humidity at 50%.

A study in 2000 on North Brother Island, New Zealand (4) reported that some tuatara (*Sphenodon guntheri*) could be induced to lay eggs using oxytocin. Nine of 21 females given oxytocin began to lay clutches of eggs (3–7 eggs/clutch) within 15–70 minutes of receiving the injection. However, eggs from two of those females were small and soft and did not develop successfully. The remaining 12 females did not respond to the oxytocin injection. In 2000, a total of 21 gravid female tuatara received an injection of oxytocin (Oxytocin-s,10 IU/mL) into the body cavity (details of total monitoring time not provided).

A replicated, controlled study in 1978–2006 in a laboratory in the USA (5) found that inducing eastern painted turtles *Chrysemys picta picta* with oxytocin did not affect hatching success when compared to eggs from natural nests, and that 13 turtle species could be induced using oxytocin and arginine vasotocin (AVT) on their own, or in combination with other drugs. Painted turtle hatching success was similar for oxytocin-induced eggs (57 of 62, 92%) and natural nest eggs (58 of 60, 97%). Across 13 turtle species, the number of turtles that laid all eggs after their first injection was 64–97% with oxytocin (0.7–4 units/100 g), 0–50% with AVT (5–50 ng/g), 33–90% with oxytocin and ketamine (<25 or 35 mg/kg), 50% with oxytocin and propranolol (14–38 μg/kg), and 57% with AVT and propranolol (11–14 μg/kg). Sixty painted turtle eggs were collected from wild nests, and 14 turtles were collected before laying and induced with oxytocin (1.4–2.5 units/100 g), yielding 62 eggs. All eggs were incubated in vermiculite. In total, 245 inductions of 13 species of turtle were carried out (1–42 individuals/species): 195 with oxytocin, 22 with AVT and 28 with a combination of drugs. Oxytocin and AVT was injected in to the abdomen and ketamine and propranolol were injected into the shoulder muscle.

A replicated, controlled study (years not provided) on four river banks in Illinois, USA (6) found that using oxytocin to induce egg-laying in red-eared sliders *Trachemys scripta elegans* did not affect the total number of eggs laid. Oxytocin-induced red-eared sliders laid similar numbers of eggs (14 eggs/turtle) to sliders that were not induced (15 eggs/turtle). Female red-eared sliders were collected from four nesting sites by one of two rivers. Twenty-four turtles were found laying natural nests. These turtles were caught, palpated to confirm that egg-laying was complete and eggs were counted in nests. Oxytocin (0.2 ml/kg) was used to induce egg laying in 241 turtles.

A replicated study in 2005–2008 in laboratory conditions in South Island, New Zealand (7) found that most wild pregnant common geckos *Hoplodactylus* *maculatus* brought into captivity were successfully induced after receiving the hormone arginine vasotocin (AVT). Most (number not given) of the 22 female geckos that received AVT delivered fully developed, viable offspring within 6 h of hormone injection. Pregnant common geckos were collected from the wild in November 2005 and October 2007. Initially, two females that were carrying fully developed embryos well beyond the expected term were induced with AVT (dissolved in 0.8% saline to deliver 150 ng/g of the hormone). A further 20 females that went beyond their expected term were induced.

A replicated, controlled, before-and-after study in 1998–2009 in Hainan Province, China (8) found that captive four-eyed turtles *Sacalia quadriocellata* began reproducing after six years after some individuals received hormone injections, but fertility and hatching success of eggs was low. Results were not statistically tested. In 2004–2008, five of 20 eggs (25%) from hormone-injected females were fertile, compared to 11 of 21 eggs (52%) from females injected with a saline solution (numbers taken from table). In 2008–2009, three of 43 eggs (7%) from females kept in outdoor pools and given no injections were fertile. In 1998, 28 female and 17 male turtles were acquired and kept in indoor pools (60 x 80 cm). In 2004–2007, eighteen females and 12 males were given luteinizing hormone-releasing hormone analogue (females: 8 μg/kg; males 4 μg/kg) and human chorionic gonadotropin (females: 1600 IU/kg; males 800 IU/kg). Hormones were injected into the hind leg muscles every 10 days up to 10 times/year. The remaining 10 females and five males were injected with a saline solution. In 2007–2008, five females and five males were moved to an outdoor pond (10 m^2), and in 2008–2009, eighteen females and 12 males were kept in the outdoor pond.

A before-and-after study in 2006–2009 in seawater tanks in Okinawa Island, Japan (9) found that administering a follicle-stimulating hormone ('FSH') to captive female hawksbill turtles *Eretmochelys imbricata* resulted in ovulation and egg formation in all individuals. Following an injection of follicle-stimulating hormone, four female hawksbill turtles ovulated and formed eggshells within 2–4 days. The authors reported that none of the turtles had ovulated in captivity before. In July 2009, four sexually mature female turtles were administered the hormone 'FSH' via intra-muscular injection (see original paper for details). Two of the turtles were wild caught in 1996–1998 and were developing follicles/considered sexually mature from 2006 and two were bred in captivity in 1994 and were considered sexually mature from 2008. All turtles were isolated for the year prior to being injected. Turtles were monitored for signs of ovulation using ultrasound.

(1) De Bruin R.W.F. & Zwartepoorte H.A. (1994) Captive management and breeding of *Cuora aurocapitata* (Testudines: Emydidae). *Herpetological Review*, 25, 58–59.

(2) Nelson N.J., Thompson M.B., Pledger S., Keall S.N. & Daugherty C.H. (2004) Egg mass determines hatchling size, and incubation temperature influences post-hatching growth, of tuatara *Sphenodon punctatus*. *Journal of Zoology*, 263, 77–87.

(3) King R.B. & Stanford K.M. (2006) Headstarting as a management tool: a case study of the plains gartersnake. *Herpetologica*, 62, 282–292.

(4) Mitchell N.J., Nelson N.J., Cree A., Pledger S., Keall S.N. & Daugherty C.H. (2006) Support for a rare pattern of temperature-dependent sex determination in archaic reptiles: evidence from two species of tuatara (Sphenodon). *Frontiers in Zoology*, 3, 1–12.

(5) Feldman M.L. (2007) Some options to induce oviposition in turtles. *Chelonian Conservation and Biology*, 6, 313–320.
(6) Tucker J.K. (2007) Comparison of clutch size from natural nests and oxytocin induced clutches in the red-eared slider, *Trachemys scripta elegans*. *Herpetological Review*, 38, 40.
(7) Cree A. & Hare K.M. (2010) Equal thermal opportunity does not result in equal gestation length in a cool-climate skink and gecko. *Herpetological Conservation and Biology*, 5, 271–282.
(8) He B., Liu Y., Shi H., Zhang J., Hu M., Ma Y., Fu L., Hong M., Wang J., Fong J.J. & Parham J.F. (2010) Captive breeding of the four-eyed turtle (*Sacalia quadriocellata*). *Asian Herpetological Research*, 1, 111–117.
(9) Kawazu I., Suzuki M., Maeda K., Kino M., Koyago M., Moriyoshi M., Nakada K. & Sawamukai Y. (2014) Ovulation induction with follicle-stimulating hormone administration in hawksbill turtles *Eretmochelys imbricata*. *Current Herpetology*, 33, 88–93.

14.17. Release captive-bred reptiles into the wild

Background

Captive breeding is normally used to provide individuals that can then be released into the wild (often called 'reintroduction') to either re-establish a population that has been lost, or to augment an existing population ('restocking').

Release techniques vary considerably, from 'hard releases' involving the simple release of individuals into the wild to 'soft releases' which involve a variety of adaptation and acclimatisation techniques before release, or post-release feeding and care.

This action includes studies describing the effects of release programmes for captive-bred reptiles that do not specifically test the effectiveness of specific release techniques. For studies that compare specific release techniques see *Use holding pens or enclosures at release site prior to release of captive-bred reptiles; Use holding pens or enclosures at release site prior to release of wild reptiles* and *Release reptiles into burrows*.

Due to the number of studies found, this action has been split by species group, though no studies were found for amphisbaenians.

Sea turtles

- **Three studies** evaluated the effects of releasing captive-bred sea turtles into the wild. Two studies were in the Gulf of Mexico[1,2] and one was in the Caribbean[3].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (3 STUDIES)

- **Reproductive success (1 study):** One replicated study in the Caribbean[3] found that eight of over 30,000 captive-bred green turtles released into the wild (around 15,000 reared to one year or more in captivity) were observed nesting and two produced clutches of >100 eggs with hatching success of 63% and 88%.
- **Survival (3 studies):** Three replicated studies in the Gulf of Mexico[1,2] and the Caribbean[3] reported that following releases of captive-bred Kemp's ridley turtles[1,2] and

green turtles[3] into the wild, 120–606 of 22,000–30,000 turtles survived for 1–19 years after release.

- **Condition (1 study):** One replicated study in the Gulf of Mexico[1] found that captive-bred Kemp's ridley turtles released into the wild grew by 19–59 cm over 1–9 years.

BEHAVIOUR (0 STUDIES)

A replicated study in 1978–1992 at several sites on the Texan coast of the Gulf of Mexico, USA (1; same experimental set-up as 2) found that some released captive-bred and reared Kemp's ridley turtles *Lepidochelys kempii* survived up to 9 years in the wild. Of the 22,608 turtles released, more than 117 were recaptured in the Gulf of Mexico and adjacent bays 1–9 years after release. Recaptured turtles grew by 19–59 cm (straight carapace length) over a period of 1–9 years. In 1978–1992, a total of 22,608 turtles were released into the Gulf of Mexico or adjacent bays, including 18,790 yearlings. Of these yearlings, 18,174 (97%) were released into the Gulf of Mexico, and 616 (3%) into adjacent bays. Turtles were recaptured on an ad-hoc basis by a sea turtle stranding and salvage network and commercial or recreational fishers.

A replicated study in 1978–1993 at 13 sites on the Mexican, Texan and Floridian coasts of the Gulf of Mexico (2; experimental set-up as 1) found that following large scale releases of captive-bred yearling Kemp's ridley turtles *Lepidochelys kempii*, some individuals survived and were recaptured 1–10 years after release. At least 606 turtles survived and were recaptured 1–2 years after release, and at least 59 survived and were recaptured 3–10 years after release. In 1978–1992, a total of 22,255 yearling turtles were released at 13 locations, with 197 released in Campeche, Mexico; 3,268 in west Florida, USA; and 18,174 in Texas, USA. Turtles were recaptured on an ad-hoc basis by a sea turtle stranding and salvage network and commercial or recreational fishers.

A replicated study from 1980–2005 in the Cayman Islands and wider Caribbean (3) found that some released captive-bred and reared green turtles *Chelonia mydas* were recaptured as adults throughout the Caribbean, and some were observed successfully nesting. A total of 392 tagged animals were recaptured at intervals of six months to 19 years after release. Of these, 160 were recaptured in the Cayman Island and 232 from elsewhere (2 from Belize, 176 from Cuba, 8 from Honduras, 1 from Mexico, 38 from Nicaragua, 2 from Panama, 4 from USA and 1 from Venezuela). Eight turtles were observed nesting, and two individuals produced clutches of 112 and 110 eggs, with hatching success of 63% and 88%. Rearing occurred at the Cayman Turtle Farm: a commercial turtle meat operation that raised green turtles from captive adults and released excess turtles in to the wild. Eggs were laid on an artificial beach, incubated in a hatchery and then hatchlings reared in groups. Between 1980 and 2001, turtles were released (16,422 hatchlings, 14,347 yearlings and 65 turtles of 19–77 months old) during October–November. Approximately 80% of all turtles released were tagged using a variety of methods (notching, flipper tags and living tags). Recapture information came from intentional and accidental capture by fisheries throughout the Caribbean, stranding networks in the USA, an active recapture effort in 1994 (Cayman Islands) and observations of nesting females.

(1) Caillouet C.W., Fontaine C.T., Manzella-Tirpak S.A. & Williams T.D. (1995) Growth of head-started Kemp's ridley sea turtles (*Lepidochelys kempii*) following release. *Chelonian Conservation and Biology*, 1, 231–234.
(2) Caillouet C.W., Fontaine C.T., Manzella-Tirpak S.A., & Shaver D.J. (1995) Survival of head-started Kemp's ridley sea turtles (*Lepidochelys kempii*) released into the Gulf of Mexico or adjacent bays. *Chelonian Conservation and Biology*, 1, 285–292.
(3) Bell C.D., Parsons J., Austin T.J., Broderick A.C., Ebanks-Petrie G. & Godley B.J. (2005) Some of them came home: the Cayman Turtle Farm headstarting project for the green turtle *Chelonia mydas*. *Oryx*, 39, 137–148.

Tortoises, terrapins, side-necked & softshell turtles

- **Fourteen studies** evaluated the effects of releasing captive-bred tortoises terrapins, side-necked & softshell turtles into the wild. Five studies were in the USA[5-7,10,13], three were in Italy[8,9,12], two were in the Seychelles[4a,4b], and one was in each of Madagascar[1], Australia[2] and Spain and Minorca[11] and one was global[3].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (13 STUDIES)

- **Abundance (1 study):** One global review[3] found that when using recruitment to the adult population as a measure of success, 32% of reptile translocations/releases (releases of captive individuals were 7% of total projects) were successful.
- **Occupancy/range (1 study):** One review in Australia[2] found that two of three releases of captive-bred Western swamp tortoises were classified as successful.
- **Reproductive success (2 studies):** Two studies (including one replicated study) in Italy[8,12] reported evidence of a gravid female[8] and successful reproduction[12] following release of captive-bred European pond turtles.
- **Survival (11 studies):** Six of nine studies (including two replicated, controlled studies) in Madagascar[1], the Seychelles[4a,4b], the USA[5,6,7,10,13] and Italy[8] reported that 66–100% of 5–80 captive-bred tortoises and turtles released into the wild survived over monitoring periods of six months to two years[1,4a,6,8,10,13]. Two studies[4b,7] reported that 16–20% of 5 and 246 individuals survived over two years. The other study[5] reported that some of over 250 individuals (number not given) were recaptured over a year of monitoring. One study[7] also found captive-bred alligator snapping turtles that were older at their time of release had higher survival than younger turtles. One replicated study in Italy[9] found that annual survival of released captive-bred European pond turtles was 67–91%. One replicated study in Spain and Minorca[11] found that survival of captive-bred Hermann's tortoises was higher after three years after release into the wild compared to 1–2 years after release. The study[11] also found that after three years, survival of released tortoises was similar to that of wild tortoises in one population, but lower in a second population.
- **Condition (2 studies):** One of two controlled studies (including one replicated study) in the USA[5,7] found that released captive-bred juvenile alligator snapping turtles grew at a similar rate and achieved higher body condition than juveniles that remained in captivity[5]. The other study[7] found that released alligator snapping turtles had similar body conditions compared to individuals that remained in captivity.

BEHAVIOUR (1 STUDY)

- **Behaviour change (1 study):** One randomized study in the USA[13] found that captive-bred Blanding's turtles released into open water habitat had larger home ranges than those released into places dominated by cattail or willows.

A replicated, controlled study in 1986–1999 in dry deciduous forest in Madagascar (1) found that released captive-bred subadult ploughshare tortoises *Geochelone yniphora* that were held in pens for four weeks prior to release and provided food and water survived at least one year in the wild. All five released captive-bred subadult ploughshare tortoises survived at least one year in the wild, and settled within 138–523 m of the release site. Released tortoises returned fewer times to the same locations over time (26% of daily locations were the same) compared to wild tortoises (44% of daily locations were the same). Over one year, daily movements were similar between released and wild tortoises (see original paper for details). A captive breeding programme was established in 1986 and five first-generation offspring (8–9 years old) were released in February 1998 and monitored using radio transmitters until January 1999. Five wild juvenile tortoises of a similar size and age in the same region were monitored at the same time. Released tortoises were placed in an acclimatisation pen for four weeks at the release site and initially provided with food and water.

A review of releases of captive individuals to wetland reserves and an island off the coast of Western Australia, Australia (2) found that two of three releases of captive-bred western swamp tortoises *Pseudemydura umbrina* were classified as successful. Two of three tortoise releases were considered successful and the success of a further release could not be determined. In 1994–2001, 12–130 tortoises were released at three sites. Animals were translocated from a zoo. The definition of successful translocation was not stated but for other species in the review it included measures of population increase and persistence.

A review of worldwide reptile translocation projects during 1991–2006 (3) found that a third of the projects, that included some releases of captive-bred animals, were considered successful with substantial recruitment to the adult population. Of the 47 translocation projects reviewed (39 species), 32% were successful, 28% failed and long-term success was uncertain for the remaining 40%. Projects that translocated animals due to human-wildlife conflicts failed more often (63% of 8 projects) than those for conservation purposes (15% of 38) and those for research purposes (50% of 5). Success was independent of the life-stage translocated/released, number of animals released and geographic region (see paper for details). Releases of captive-bred animals made up 7% of the projects, and individuals involved were adults in 75% of cases, juveniles and sub-adults in 64% of cases and eggs in 4% of cases. The most common reported cause of failure was homing and migration with the second most common reported cause being insufficient numbers, human collection and food/nutrient limitation all equally reported. Success was defined as evidence of substantial recruitment to the adult population during monitoring over a period at least as long as it takes the species to reach maturity.

A study in 1997–2011 on tropical islands in the Seychelles (4a) found that captive-bred Arnold's giant tortoises *Dipsochelys dussumieri arnoldi* released into the wild survived for at least five years. After being released from captivity on

Silhouette Island, five of five Arnold's giant tortoises survived in the wild for at least five years and nests were found one and five years after release. After the remaining giant tortoises from a captive-breeding programme were released in 2011, the author reported that the tortoises settled in their release environments. In 1997–2011, a giant tortoise captive breeding programme was carried out on Silhouette Island. In 2006, five adult Arnold's giant tortoises (three male and two female) were released at Grande Barbe, Silhouette Island (2010 ha). After the programme closed in 2011, 38 juvenile Arnold's giant tortoises were moved and released on North Island (210 ha) in February–March 2011 and 92 were released on Fregate Island (207 ha).

A study in 1997–2011 in Silhouette Island, Seychelles (4b) found that captive-bred adult black mud turtles *Pelusios subniger parietalis* released into the wild survived at least six months to two years and yellow-bellied mud turtles *Pelusios castanoides intergularis* survived at least several months. In 2002, five of five adult black mud turtles survived at least six months. One of the adult black mud turtles was recaptured two years after release. The author reported that released captive-bred and captive-maintained yellow-bellied mud turtles were seen regularly after release in 2010. In 2002, five captive-bred adult black mud turtles were released at Grande Barbe, Silhouette Island. In 2003, eighteen captive-bred juvenile turtles were released and in 2009 the remaining adult black mud turtles from the captive breeding programme were also released (total number of adults not provided). Three captive-bred juvenile and several captive-maintained adult yellow-bellied turtles were released into a lake in 2010 (total number of adults not provided). Adult black mud turtles were monitored by occasional trapping and tracking using radio tags for six months.

A controlled study in 2007–2008 along a river in southern Oklahoma, USA (5; same experimental set-up as 6) found that some captive-bred alligator snapping turtles *Macrochelys temminckii* released into the wild were recaptured in the year following release. Following release of 16 juveniles, individuals were recaptured on 5 occasions in the year of release and on 18 occasions the year after release (number of individuals recaptured not given). Individuals from a group of translocated adults were recaptured on 50 occasions (249 released, number of individuals not given). Released juveniles grew at a similar rate to those that remained in captivity (released: 0.07 mm/day, captive: 0.09 mm/day), but obtained higher body condition (data reported as statistical model result). Sixteen captive-bred juveniles were release at one location in June 2007, and a further 26 juveniles remained in captivity. An additional 249 adult turtles were confiscated from a turtle farm and released in groups of 27–62 at seven pools adjacent to the river in April 2007. Turtles were recaptured with baited hoop nets in May–August 2007 and 2008.

A study in 2007–2008 in along a river in Oklahoma, USA (6; same experimental set-up as 5) found that following release into the wild, most juvenile captive-bred alligator snapping turtles *Macrochelys temminckii* and subadults recovered from a turtle farm survived at least one summer in the wild. After one year, 24 of 32 (75%) released alligator snapping turtles were still alive. Two turtles were lost within 45 days of release and a further six turtles were lost by the beginning of the second year in the wild (see original paper for details). Captive-bred juveniles dispersed similar distances after release (765 m) as released subadults from turtle farms (769 m), but had smaller home ranges (captive-bred: 730 m; turtle farm: 1,789 m). In June 2007, sixteen captive-bred juvenile turtles (bred and reared in the Tishomingo National Fish Hatchery; born in 2002 or 2004) were radio-tagged and released into a single site (a river reach adjacent to the Washita River). In 2006, sixteen subadult alligator snapping turtles recovered from a turtle farm were radio-tagged and released at the same location. A further 250 turtles from the turtle farm were released at six locations in 2007 (monitoring data not provided). Turtles were monitored weekly during the summer months in 2007 and 2008.

A replicated, controlled study in 2008–2012 in two rivers in Oklahoma, USA (7) found that releasing captive-bred alligator snapping turtles *Macrochelys temminckii* resulted in some individuals surviving at least four years in the wild. At least 40 of 246 turtles (actual number not given) were recaptured at least once 1–4 years after release, and overall annual survival was estimated at 59%. Turtles that were older at their time of release were estimated to have higher annual survival than younger turtles (5 years old: 100%; 4 years old: 70%; 3 years old: 59%). Recaptured turtles all showed increases their shell size compared to when released (average of 7–29% growth/year). Body condition of released turtles was similar to that of turtles that remained in captivity (reported as statistical model output). In 2008–2010, a total of 246 turtles were released into two rivers. Turtles were captive-bred and raised in captivity for 3–7 years. Annual trapping was carried out in 2008–2012 during May–August for 60–189 trap nights/year. Size of recaptured turtles (number not given) was compared to 224 still in captivity.

A study in 2003–2009 in a wooded wetland in northern Italy (8) found that a population of juvenile captive-bred European pond turtle *Emys orbicularis galloitalica* hatchlings released into an area where predators were removed and excluded survived in the wild for at least two years and bred. Ten of 12 (83%) nine-month-old captive-bred European pond turtles survived at least two years in the wild. Five years after the first releases, the first female turtle was confirmed to be carrying eggs. Authors reported that the two turtles that died in the first year after release were probably predated by terrestrial mammals. In September 2003 (eight individuals) and 2005 (four individuals) hatchling European pond turtles born in captivity (sourced from a private breeder) were reared in a terrarium for eight months. In May 2004 and 2006, juvenile turtles were moved to a predator-proof acclimatisation cage (1 x 2 m) for one month prior to release into a fenced pond inside a fenced 1 ha wetland complex in a regional park (see original paper for details). A resident population of largemouth bass *Micropterus salmoides* was controlled prior to release by catching with pole and line. Larger fish predators were excluded from shallow waters in the release pond using fences woven from branches. A sand and dirt nesting area (2.5 m high x 15 m long) was created in the release area.

A replicated study in 2008–2015 in three locations on a river in Liguria, Italy (9) found that three populations of released captive-bred European pond turtles *Emys orbicularis* survived in the wild for at least 8 years. After eight years, 80 of 200 (40%) captive-bred released European pond turtles were estimated to still be alive in three different sites. Annual survival rates of captive-bred turtles released were 67–91% (survival rates differed between release sites, see original paper for details). In 2000–2015, five-hundred captive-bred pond turtles were hatched in an outdoor breeding facility. Hatchlings were reared in an aquarium for two years before being returned to the breeding facility for outdoor acclimatization (duration not specified) prior to release. Approximately 60% of hatchlings survived 3–4 years in captivity. Captive-bred turtles were released annually in three sites in June–July in 2008–2015 (200 individuals released). Survival rates were estimated based on three trapping surveys carried out for three days at a time, in May–August 2008–2015.

A replicated study in 2014–2015 in four desert scrub vegetation sites in Nevada, USA (10) found that more than half of released captive-bred juvenile Mojave desert tortoises *Gopherus agassizzii* survived at least six months and settled into home ranges within two months of release. Six months to one year after release, 53 of 80 (66%) released captive-bred juvenile desert tortoises were still alive. The authors reported that of 25 known tortoise deaths, 14 were due to starvation or exposure, and the remaining 11 showed signs of predation or scavenging. Overall, 46 of 71 (65%) desert tortoises settled into a home range pattern within two weeks, all 71 had settled within two months, and nine died before establishing a home range. In September 2014 and April 2015, eighty desert tortoises were hatched and reared in captivity (ages ranged from 6 months to 4 years) and released into four different locations in the Mojave Desert (19–21 tortoises released/locations). Tortoises were released at least 20 m apart and radio tracked weekly during March–October and every two weeks during November–February from release until September 2015. Two tortoises lost their transmitters and were excluded from survival numbers.

A replicated study in 1987–2015 in sand dune and mixed forest habitats in Spain and Minorca (11) found that many captive-maintained, released Hermann's tortoises *Testudo hermanni hermanni* survived and that survival rates of released tortoises increased three years after release. During the first two years after release, average survival rates of translocated tortoises were 44–90% in Spain (66 individuals) and 79–85% in Minorca (48 individuals). In the third year after release, survival rates of translocated tortoises in Spain (98%) were similar to wild-born tortoises (98%) but survival rates of translocated tortoises in Minorca (89%) were lower than wild-born tortoises (97%). Body condition of tortoises before release did not affect whether or not a tortoise was subsequently found dead or alive (see original paper for details). Tortoises were maintained in captivity, though their origin and total time in captivity was unknown. In Spain, 66 captive tortoises were released into a protected reserve in four batches: September 1987, May–August 1988, March 1997 and September 1998. In Minorca, 48 tortoises were released in March–April 2004. The amount of time tortoises had spent in captivity prior to release was unknown. Tortoises were monitored for 4–10 days a year (Spain: 28 years; Minorca: 14 years) using capture-mark-recapture surveys.

A replicated study in 2016–2017 in two semi-permanent clay ponds in Savona, north-west Italy (12) found that released captive-bred European pond turtles *Emys orbicularis ingauna* bred in the wild and that after release they had adapted their diets to eat food that was available to them. Three turtle nests with successfully hatched eggs were observed at one of the release sites in 2017 (year of release not provided). Captive-bred released turtles ate a range of invertebrates, seeds and plant matter in the wild, although they had been fed commercial shrimp pellets and frozen shrimp and fish in captivity (see original paper for details). Turtles were bred and reared in captivity from 1999 for 2–5 years prior to their release into two sites (year of release not stated in the original paper). Dietary analysis was carried out on droppings from 25 released turtles that were recaptured in June 2016.

A randomized study in 2014–2016 in a wetland complex in Michigan, USA (13) found that all released captive-bred Blanding's turtles *Emydoidea blandingii* survived their first winter hibernation and most survived at least one year after release. All 24 turtles survived to the spring after release (approximately 9 months) and at least 16 turtles survived for 17 months (only one turtle death was confirmed by the presence of a carcass). Survivorship was lower for turtles released into open water habitat (best case: 5 of 6 turtles survived, worst case: 2 of 6 turtles survived) compared to turtles released into areas vegetated with cattail *Typha* sp. or willow *Salix* sp. (best case: 18 of 18 turtles survived, worst case: 13 of 18 turtles survived; no statistical tests were carried out). Average home ranges were larger for turtles released into open water habitat (2.9 hectares) compared to turtles in cattail- or willow-vegetated habitat (0.4–0.6 hectares). In total 24 individuals were selected randomly from turtles bred, hatched and reared in captivity. Turtles were at least one year old and shell length was >10.2 cm. Turtles were released in June 2014 into four wetland habitats (6 individuals per group): open water, sparse cattail vegetation, dense cattail vegetation and willow (see original paper for details). Turtles were monitored by radio transmitter in June 2014–November 2015 (for 515 days in total) once a week during May-September and every two weeks during October–April. Turtles were recaptured autumn 2014, spring 2015, and autumn 2015 to replace/remove transmitters. Best case survival estimates are based on known mortality, worst case include turtles whose radio transmitters were lost or failed and turtles were presumed dead.

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Snakes & lizards

- **Ten studies** evaluated the effects on reptile populations of releasing captive-bred snakes and lizards into the wild. Three studies were in New Zealand[7,9,10], two were in the USA[1,2] and one was in each of the Galápagos[3], Spain[5], Australia[6] and Canada[8] and one was global[4].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (10 STUDIES)

- **Abundance (2 studies):** One global review[4] found that when using recruitment to the adult population as a measure of success, 32% of reptile releases (releases of captive individuals were 7% of total projects) were successful. One review in New Zealand[10] found that 13% of lizard releases (some involving captive-bred animals) found evidence of populations growth
- **Reproductive success (3 studies):** Three studies (including two reviews) in the USA[1] and New Zealand[7,10] found evidence of breeding following release in one of two captive-bred populations of cornsnakes[1], one captive-bred population of Otago skinks[7] and in at least 16 lizard mitigation translocations[10], some of which involved captive-bred animals[10].
- **Survival (9 studies):** One replicated, controlled study in Spain[5] found that released large psammodromus lizards had similar annual survival compared to resident lizards. Two of six studies (including one replicated study and two reviews) in the USA[1,2], Australia[6], New Zealand[7,10] and Canada[8] reported that 13% of 40 indigo snakes were re-sighted at least once during 5–8 years following release[2] or that 58% of 12 Otago skinks survived at least 18 months[7]. Two studies[6,8] found that zero of nine[6] and 27[8] individuals survived more than 143 days[6] or beyond their first hibernation[8]. The other two studies[1,10] found that one of two[1] and five of 53[10] releases (only some of which involved captive-bred animals[10]) failed completely (no individuals survived). One study in New Zealand[9] found that survival of captive-bred Otago skinks released into an enclosure was higher when mice had been eradicated compared to when skinks were released in the presence of mice. One replicated study in the Galápagos[3] found that while releases were ongoing over a decade (183 released in total), 17–32 Galápagos land iguanas[3] were recaptured each year**.**
- **Condition (1 study):** One controlled study in New Zealand[7] found that body condition of captive-reared Otago skinks was higher than wild skinks, but sprint speed was lower.

BEHAVIOUR (1 STUDY)

- **Behaviour change (1 study):** One replicated, controlled study in Spain[5] found that released large psammodromus lizards moved between habitat fragments more frequently than resident lizards but showed similar behaviour in three other measures.

A review in 1991 of reptile translocation and release programmes in New Jersey, USA (1) found that of two releases of captive-bred, newly born cornsnake *Elaphe guttata*, one population survived at least four years, while the other survived one–two years. In one site, 17 of 158 (11%) released newly born captive-bred cornsnakes survived at least one year and six released snakes survived at least 4 years and bred. In a second site, six of 262 (2%) released snakes were recaptured one year after release and none were recaptured two years after release. In the first site, 158 newly born captive-bred cornsnakes were released into a known hibernaculum in 1982–1988. In the second site, 262 newly born captive-breds were released in 1985–1989.

A study in 1980–2001 on an island off the coast of Florida, USA (2) found that a small number of released (some captive-bred) eastern indigo snakes *Drymarchon couperi* survived 5–8 years in the wild. In the 17–20 years after 40 eastern indigo snakes were released, five snakes were recorded in the wild and the last snake was observed 5–8 years after release (1983: 1 individual; 1985: 1 individual; 1986: 2 individuals; 1988: 1 individual). In 1980–1982, forty eastern indigo snakes (hatchlings and juveniles from a captive breeding colony, wild-caught adults, confiscated snakes and donated from zoos) were released onto St Vincent Island National Wildlife Refuge (51 km^2). Snakes were monitored using combinations of cameras in gopher tortoise *Gopherus polyphemus* burrows and drift fence/pitfall trap arrays in autumn, winter and spring 1983–1990, January and December 2000, and April 2001. Sightings (unverified) were also recorded but are not reported here.

A replicated study in 1991–1993 on a tropical island in the Galápagos, Ecuador (3) found that following release of captive-bred Galápagos land iguanas *Conolophus subcristatus*, some survived and reproduced. Between 17–32 iguanas were recaptured/year. More offspring of released iguanas were captured after most cats *Felis catus* were eradicated from the island (1 and 14 adults and 6 and 14 sub-adults and juveniles/year) than before the cat control program began (1 and 0 adults and 6 and 4 sub-adults and juveniles/year). In 1991–2003, a total of 183 captive-bred iguanas were released over six releases (15–63 released every 1–5 years). Cat eradication started in 2001 and was completed in 2003. Iguanas were surveyed (6 days in June–July) before (1999–2000) and after (2002–2003) the majority of cat eradication had been completed.

A review of worldwide reptile translocation projects during 1991–2006 (4) found that a third of the projects, that included some releases of captive-bred animals, were considered successful with substantial recruitment to the adult population. Of the 47 translocation projects reviewed (39 species), 32% were successful, 28% failed and long-term success was uncertain for the remaining 40%. Projects that translocated animals due to human-wildlife conflicts failed more often (63% of 8 projects) than those for conservation purposes (15% of 38) and those for research purposes (50% of 5). Success was independent of the life-stage translocated/released, number of animals released and geographic region (see paper for details). Releases of captive-bred animals made up 7% of the projects, and individuals involved were adults in 75% of cases, juveniles and sub-adults in 64% of cases and eggs in 4% of cases. The most common reported cause of failure was homing and migration with the second most common reported cause being insufficient numbers, human collection and food/nutrient limitation all equally reported. Success was defined as evidence of substantial recruitment to the adult population during monitoring over a period at least as long as it takes the species to reach maturity.

A replicated, controlled study in 2001–2006 in two sites of forest fragments among cereal field in northern Spain (5) found that released captive-bred large psammodromus lizards *Psammodromus algirus* had similar survival compared to resident lizards, and a newly established population persisted for at least four years. Survival for a year after release was similar for released captive-bred lizards (2001–2002: 26 of 178, 15%; 2002–2003: 19 of 187, 10%) and residents (2002–2003: 4 of 30, 13%). A release site where only three native lizards were capture in 2002 still hosted a population in in 2006 (at least 6 individuals). More captive-bred lizards moved between habitat fragments (8 of 48 lizards) than did residents (2 of 112 lizards), though three other measures of movement and activity were similar between captive-bred and resident lizards (see paper for details). Captive-bred lizards were released in groups of 5–7 in two woodland fragments (0.9–5.2 ha) in each of two sites (located 8 km apart, two fragments/site). One site had a viable resident lizard population while the other did not. Released and resident lizards were monitored in spring and summer by walking around study sites and adjacent areas and noosing all detected lizards in 2002, 2003 (15 days each) and 2006 (two days).

A study in 2007 in a site of mixed sand dunes, *Acacia* spp. and shrubland in South Australia (6) found that releasing captive-bred woma pythons *Aspidites ramsayi* into a large, fenced enclosure was unsuccessful due to predation. All pythons died between 41 and 123 days after release, all most likely due to attack or predation by mulga snakes *Pseudechis australis*. Two snakes had lost weight (10–37% of release mass) but were not considered emaciated. Nine captive-bred sibling pythons (two females, seven males, hatched in 2002) had radio transmitters surgically implanted in April 2007. They were released in September 2007 (weighing 890–1,350 g) into a large enclosure (60 km^2) free of non-native mammalian predators. The snakes were from a wild stock originating from close to the release site. Four snakes were released into a release-pen, but all escaped within two months. Pythons were located daily until death.

A controlled study in 2009–2011 in one site of temperate shrubland in Alexandra, New Zealand (7) found that releasing captive-reared Otago skinks *Oligosoma otagense* into a fenced enclosure resulted in some surviving for at least 18 months and some breeding successfully. Most captive-bred skinks survived for at least 12–18 months after release (12 months: 75%, 9 of 12 skinks survived, 3 females disappeared; 18 months: 58% survival, 2 males disappeared). Three newborn young were recorded two years after release. Body condition of captive-reared skinks was higher than wild skinks (various species of *Oligosoma* skinks; presented as condition index), but sprint speed was lower (captive-bred: 0.4–0.6 m/s; wild: 0.9–1.5 m/s). In November 2009, twelve skinks from captive stock (five males, seven females, at least 3rd generation captive-born) were released. The release site (0.3 ha) was surrounded by a 1.9 m high mammal-proof fence and was free of all mammals for five months prior to release. Post-release visual monitoring using a camera to photograph and identify all lizards seen at sunning spots (rocks) was performed for two hours, 1–5 times/month from November 2009 to May 2011 (43 searches). Sprint speed was measured for skinks in captivity (29 skinks) and those in the wild (93 skinks).

A replicated study 2006 in a nature reserve within a wider urban setting in Ontario, Canada (8) found that captive born massasauga rattlesnakes *Sistrurus catenatus catenatus* released into the wild did not survive hibernation. Following release, at least 19 of 27 (70%) of rattlesnakes survived 19 weeks to hibernation (three died from predation, three transmitters failed, one died from human attack, one died from unknown causes). No rattlesnakes survived the hibernation period (10 died from exposure on the surface, four died from predation, four died from unknown causes, one was killed by human attack). In 2003, two gravid female rattlesnakes were rescued from a development site in Windsor, Ontario, Canada and their young raised in captivity. In 2006, the 3-year-old snakes (27 individuals) were implanted with radio transmitters and released into a nature reserve which had a natural population of rattlesnakes until at least the mid-1970s. Snakes were tracked daily for the first two weeks after released and then fortnightly thereafter.

A study in 2009–2012 in an area of mixed shrub and grassland in Otago, New Zealand (9) found that survival of captive-bred Otago skinks *Oligosoma otagense* released into an enclosure was higher for those released when house mice *Mus musculus* had been eradicated compared to when skinks were released in the presence of mice. Authors reported that post-release survival was higher for skinks released with no mice present (44%) compared to survival of skinks released just prior to reinvasion by mice (15%; see paper for details). Survival of established skinks (2 years after their release) after the mouse reinvasion was higher (91%) than for newly released skinks in the presence of mice (17%). In 2009, a 0.3 ha area was enclosed within a mammal resistant fence (1.9 m high), and 12 captive-bred adult skinks were released in the enclosure following eight weeks in quarantine. In 2011, an additional 16 skinks were quarantined and released. Over a six-month period prior to the release, all mammals inside the enclosure were eradicated using a range of baited traps. House mice reinvaded during 2012 and were again eradicated using live capture traps and poison bait stations. In 2009–2012, starting 7–10 days after release, skinks were monitored every 15 days by a walking survey of the enclosure.

A review published in 2016 of lizard translocation and release projects (some involving captive-bred animals) in New Zealand during 1988–2013 (10) found that most projects found evidence of breeding following release, but few found evidence of population growth. Forty-five of 53 (85%) translocations/releases motivated by conservation had some post-release monitoring. Seven found evidence of population growth (more lizards found than released), 33 found that populations were smaller than the number released, at least 16 found evidence of breeding after release, and five resulted in complete failure (no lizards found). One translocation (of *Oligosoma infrapunctatum*) was later discovered to be to a location outside the species historic range. Some translocations/releases involved wild animals and others captive bred animals (project success vs source of animals not stated). Published and unpublished literature were searched, and key people associated with each project were identified and contacted for further information. Translocations/releases were considered to be motivated by conservation if the primary focus was to benefit the species or recipient site.

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Crocodilians

- **Four studies** evaluated the effects of releasing captive-bred crocodilians into the wild. Two studies were in China[3a,3b], one was in South Africa[2] and one was a global review[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (4 STUDIES)

- **Abundance (2 studies):** One global review[1] found that when using recruitment to the adult population as a measure of success, 32% of reptile translocations/releases (releases of captive individuals were 7% of total projects) were successful. One study in South Africa[2] reported that following releases of captive-bred Nile crocodiles, wild populations increased in size over 30 years, but then declined in the subsequent 15 years.
- **Reproduction (2 studies):** Two studies (one replicated) in China[3a,3b] reported that breeding or nesting was observed within four years of releasing captive-bred Chinese alligators.
- **Survival (1 study):** One study in China[3b] reported that of nine captive-bred Chinese alligators, three survived for nine years and six survived for at least one year following release.

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One replicated study in China[3a] reported that after 10 years of releases of captive bred Chinese alligators to an area that had historically been occupied, 56% of constructed ponds were occupied.

A review of worldwide reptile translocation projects during 1991–2006 (1) found that a third of the projects, that included some releases of captive-bred animals, were considered successful with substantial recruitment to the adult population. Of the 47 translocation projects reviewed (39 species), 32% were successful, 28% failed and long-term success was uncertain for the remaining 40%. Projects that translocated animals due to human-wildlife conflicts failed more often (63% of 8 projects) than those for conservation purposes (15% of 38) and those for research purposes (50% of 5). Success was independent of the life-stage translocated/released, number of animals released and geographic region (see paper for details). Releases of captive-bred animals made up 7% of the projects, and individuals involved were adults in 75% of cases, juveniles and sub-adults in 64% of cases and eggs in 4% of cases. The most common reported cause of failure was homing and migration with the second most common reported cause being insufficient numbers, human collection and food/nutrient limitation all equally reported. Success was defined as evidence of substantial recruitment to the adult population during monitoring over a period at least as long as it takes the species to reach maturity.

A study in 1967–2009 along two rivers and associated floodplains in KwaZulu-Natal, South Africa (2), found that after releasing captive-bred Nile crocodiles *Crocodylus niloticus*, the number of crocodiles counted in the wild increased over 30 years, but then began to decline. Results were not statistically tested. In the 1990s, thirty years after a programme to breed and release Nile crocodiles began, the crocodile population numbered 937–1066 individuals, compared to 344–351 individuals in the 1970s. In 2009, fifteen years later, the population numbered 128–846 individuals and the authors reported that it may have been declining after peaking in the 1990s. In January 1967–November 1974, a captive-breeding programme produced, reared and released 1,257 Nile crocodiles into a game reserve (10,000 ha). Crocodile abundance was monitored on two river systems using aerial surveys (carried out by helicopter or airplane) in 1971–1973, 1985–1986, 1989–1990, 1992–1994 and 2009. Results reported here were corrected for differences between survey methods (see original paper for details).

A replicated study in 2006–2016 in an area of ponds and dense vegetation in Anhui Province, China (3a) found that after 10 years of releases of captive-bred Chinese alligators *Alligator sinensis*, alligators occupied over half of ponds in the area, and successful reproduction was occurring. Alligators were found in 28 of the 50 ponds (56%). Survivorship of released alligators was thought to be >85% (no formal analysis carried out). Successful reproduction was recorded two years after the first release (158 eggs, producing 80 hatchlings were discovered), though the full extent of nesting was unknown. Fifty ponds (30 ha total water area) were constructed in the release area, at a cost of around $US10,000 to construct and prepare the average-sized pond. Ponds were established with terrestrial (e.g. bamboo) and aquatic vegetation, and "seeded" with fish, amphibians, and snails. Prior to release, adult alligators were isolated for 3–4 months for health screening.

In 2006–2016, eleven releases (during May–June) of 93 alligators were carried out (sex ratio 1 male:2 females). Population monitoring was carried out using spotlight surveys.

A study in 2007–2016 in a wetland in Shanghai Province, China (3b) reported that some released captive-bred Chinese alligators *Alligator sinensis* survived for 1–9 years and successfully reproduced. Three of six alligators survived for 9 years, and a further six survived at least one year following release. Nesting was reported in four years following release. In 2016, the population consisted of nine adults (released individuals), at least four wild born adults (offspring of released alligators) and around five juveniles/sub-adults. In 2007, six captive-bred alligators were released into a wetland park. In 2015–2016 a further six were released.

(1) Germano J.M. & Bishop, P.J. (2009) Suitability of amphibians and reptiles for translocation. *Conservation Biology*, 23, 7–15.
(2) Calverley P.M. & Downs C.T. (2014) Population status of Nile crocodiles in Ndumo Game Reserve, KwaZulu-Natal, South Africa (1971–2012). *Herpetologica*, 70, 417–425.
(3) Manolis C., Shirley M., Siroski P., Martelli P., Tellez M., Meurer A. & Merchant M. (2016) *CSG Visit to China, August 2016*. IUCN-SSC Crocodile Specialist Group.

Tuatara

- **Two studies** evaluated the effects of releasing captive-bred tuatara into the wild. One study was in New Zealand[2] and one was a global review[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Abundance (1 study):** One global review[1] found that when using recruitment to the adult population as a measure of success, 32% of reptile translocations/releases (releases of captive individuals were 7% of total projects) were successful.
- **Condition (1 study):** One study in New Zealand[2] found that tuatara reared close to the release site had higher growth, but similar body condition compared to individuals reared in a warmer climate.

BEHAVIOUR (1 STUDY)

- **Behaviour change (1 studies):** One study in New Zealand[2] found that tuatara reared close to the release site had similar home range sizes and post-release dispersal compared to individuals reared in a warmer climate.

A review of worldwide reptile translocation projects during 1991–2006 (1) found that a third of the projects, that included some releases of captive-bred animals, were considered successful with substantial recruitment to the adult population. Of the 47 translocation projects reviewed (39 species), 32% were successful, 28% failed and long-term success was uncertain for the remaining 40%. Projects that translocated animals due to human-wildlife conflicts failed more often (63% of 8 projects) than those for conservation purposes (15% of 38) and those for research purposes (50% of 5). Success was independent of the life-stage translocated/released, number of animals released and geographic region (see paper for details). Releases of captive-bred animals made up 7% of the projects, and individuals involved were adults in 75% of cases, juveniles and sub-adults in 64% of cases and eggs in 4% of cases. The most common reported cause of failure was homing and migration with the second most common reported cause being insufficient numbers, human collection and food/nutrient limitation all equally reported. Success was defined as evidence of substantial recruitment to the adult population during monitoring over a period at least as long as it takes the species to reach maturity.

A study in 2012–2013 in regenerating temperate forest in South Island, New Zealand (2) found that most released captive-reared tuatara *Sphenodon punctatus* survived at least 9 months after being released into a predator-free fenced enclosure with artificial burrows. After 3–5 months, all tuatara captive-reared locally (100%) and almost all tuatara captive-reared to the north of the release site (96%) survived. After 9–11 months, survival rates of tuatara reared north of the release site (70%) were higher than and locally-reared and released tuatara (67%, results were not statistically tested). Growth rates of locally-reared tuatara (0.05 mm/day) were faster than those reared away from the release site (0.04 mm/day). Changes in body condition, post-release dispersal and home range sizes were similar between locally-reared and distant-reared tuatara (see original paper for details). Juvenile tuatara (41 individuals) originating from the same wild population were released into a reserve in November–December 2012: captive-reared locally to the release site (13 individuals), and captive-reared 480 km north of the release site in a warmer climate (28 individuals). Captive-reared tuatara were hatched from artificially incubated eggs and reared until 4–6 years old. The reserve was surrounded by predator-resistant fencing and mammalian predators were mostly eradicated by 2008. Artificial burrows were buried in the release area. Tuatara were monitored by radio-tracking for 5 months (6 locally-reared, 10 north-reared individuals) and recapture surveys (all tuatara were PIT tagged) for up to 27 months after release.

(1) Germano J.M. & Bishop, P.J. (2009) Suitability of amphibians and reptiles for translocation. *Conservation Biology*, 23, 7–15.

(2) Jarvie S., Senior A.M., Adolph S.C., Seddon P.J. & Cree A. (2015) Captive rearing affects growth but not survival in translocated juvenile tuatara. *Journal of Zoology*, 297, 184–193.

14.18. Use holding pens or enclosures at release site prior to release of captive-bred reptiles

- **Two studies** evaluated the effects on reptile populations of using holding pens or enclosures at release sites prior to release of captive-bred reptiles. Both studies were in the USA[1,2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Survival (2 studies):** Two controlled studies (including one replicated study) in the USA[1,2] found that survival of captive-bred smooth green snakes[1] and desert tortoises[2] held in pens before release was similar over 3–5 months[1] or 2–3 years[2] compared to individuals released directly.

BEHAVIOUR (1 STUDY)

- **Behaviour change (1 study):** One controlled study in the USA[1] found that movement of smooth green snakes held in pens before release was similar compared to snakes that were released directly.

Background

Holding pens or enclosures at release sites (sometimes termed 'soft release') may be used to enable reptiles to become accustomed to new surroundings before release and may contain some natural habitat and burrows. Pens or enclosures may increase the chance that released animals will settle at the release site, potentially increasing the chance that the release will be successful. Captive-bred, naïve animals in particular may benefit from the acclimation period that holding pens provide.

This action discusses studies that test the effectiveness of placing captive-bred individuals into holding pens at the release site prior to release. See also: *Use holding pens or enclosures at release site prior to release of wild reptiles*.

A controlled study in 2010–2011 in a grassland site in Illinois, USA (1) found that captive-bred smooth green snakes *Opheodrys vernalis* released into holding pens before release ('soft release') had a similar chance of recapture as those released directly, and moved less than wild residents. Soft-released snakes were recaptured a similar number of times (13 recaptures/snakes) compared to hard-released snakes (6 recaptures/snake) over 3–5 months following release. Soft-released snakes moved less than residents (soft-released: 2 m/day; residents: 5 m/day), but movement of hard-released snakes (2 m/day) was similar to both soft-released and resident snakes. Eighteen captive-bred and reared snakes (≥9 g) were released in 2011 via soft-release (9 snakes; released in to 1.2 x 1.2 x 1.2 enclosure, held for 3 weeks before final release) or hard-release (9 snakes, released directly). Monitoring was completed by radiotracking and checking under coverboards on the ground at least 5 days/week for the first week, then 3 times/week for 3–5 months. Snake growth was also monitored, but only in captivity.

A replicated, controlled study in 2005–2008 in desert scrubland in California, USA (2) found that first-year survival rates of head-started released juvenile desert tortoises *Gopherus agassizii* were similar regardless of whether holding pens were used and that overall one third of head-starters survived at least three years in the wild. First-year survivorship of tortoises initially released into holding enclosures was similar (9 of 12, 75% tortoises survived) compared to those that were direct-released into the same sites (12 of 15, 80% tortoises survived). Overall survivorship of released head-started juvenile desert tortoises was 32% over three years (17 of 53 tortoises survived). In the first year after release, 42 of 53 (81%) tortoises survived, in the second year after release 32 of 42 (76%) tortoises survived and in the third year after release 17 of 32 (53%) tortoises survived. Survivorship also was similar between tortoises released in the autumn compared to the spring (see original paper for details). In autumn 2005, twelve head-started tortoises were initially placed in temporary predator-proof enclosures in three sites (4 tortoises/site), 15 head-started tortoises were direct-released in the same three sites (5/site), and a further 16 head-started tortoises were direct-released in a fourth site. In spring and autumn 2006, ten further head-started tortoises were released into the fourth site. Tortoises housed in predator-proof enclosures (each 45 m^2) were enclosed from September 2005–January 2006. All tortoises were radio-tracked weekly-biweekly during active seasons and monthly during inactive seasons from release until autumn 2008 (up to three years). Tortoises were recaptured twice/year while radio tracked for a health check.

(1) Sacerdote-Velat A.B., Earnhardt J.M., Mulkerin D., Boehm D. & Glowacki G. (2014) Evaluation of headstarting and release techniques for population augmentation and reintroduction of the smooth green snake. *Animal Conservation*, 17, 65–73.

(2) Nagy K.A., Scott Hillard L., Tuma M.W. & Morafka D.J. (2015) Head-started desert tortoises (*Gopherus agassizii*): Movements, survivorship and mortality causes following their release. *Herpetological Conservation and Biology*, 10, 203–215.

14.19. Head-start wild-caught reptiles for release

Background

Head-starting is a specialized management technique that raises early-stage reptiles (eggs, hatchlings and/or juveniles) to later life stages (juvenile, sub-adult or adult) in captivity before releasing them into the wild. Rearing animals beyond their most vulnerable stages may increase the chance of survival following release, and as such improve the chances of reintroduction success.

Here we only include those studies where eggs or juveniles were collected from the wild; for those that were bred in captivity see *Breed reptiles in captivity* and *Release captive-bred reptiles in to the wild*. See also *Release reptiles born/hatched in captivity from wild-collected eggs/wild-caught females without rearing*.

Due to the number of studies found, this action has been split by species group, though no studies were found for amphisbaenians.

Sea turtles

- **Seven studies** evaluated the effects of head-starting wild-caught sea turtles for release. Two studies were in the Caribbean Sea[1,3] and one was in each of the Torres Strait, northern Australia[2], the Gulf of Mexico[4], Japan[5], the USA[6] and Thailand[7].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (6 STUDIES)

- **Abundance (1 studies):** One replicated, before-and-after study in the USA[6] found that over the course of a 37-year head-start programme, the number of kemp's ridley nests laid on the Texas coastline increased from near zero to 119.
- **Reproductive success (2 studies):** Two studies (including one replicated, before-and-after study) in Mexico[4] and the USA[6] found that all 11 head-started Kemp's ridley turtles bred in the wild following release[4] and head-started turtles that were allowed to crawl to

the sea before recapture began laying nests on their beach of origin 10–12 years after release.

- **Survival (4 studies):** One of four studies (including two replicated and two controlled studies) in the Caribbean Sea[1], Torres Strait near Australia[2], Gulf of Mexico[4] and Japan[5] reported that all 11 head-started Kemp's ridley turtles survived at least 11–19 years following release[4]. Two of the studies[1,2] reported that 1–16% of sea turtles were recaptured 10–27 month[1] or 0.5–13 months[2] following release. The other study[5] found that four head-started hawksbill turtles survived at least 4–9 days, and one survived at least 10 months following release.
- **Condition (1 study):** One replicated study in Thailand[7] found mixed effects of tank depth on growth rate, size and body condition of green turtles during a head-starting programme and no effect of feed type.

BEHAVIOUR (2 STUDIES)

- **Use (1 study):** One replicated study in the Caribbean Sea[1] reported that one head-started green turtle travelled 2,300 km from its release location, whereas other recaptures were within 1–14 km of the release site.
- **Behaviour change (1 study):** One replicated study in the Caribbean Sea[3] found mixed effects on swimming behaviour of released head-started loggerhead turtles at 1.5 years old compared to 2.5 years old.

A replicated study in 1967–1974 in pelagic waters in the Caribbean Sea near Bermuda (1) found that some head-started and some accidentally-caught immature green turtles *Chelonia mydas* survived at least several months after release in the wild. In total, 16 of 108 (15%) released head-started or accidentally-caught immature green turtles were recaptured. Nine turtles were recovered within 10 months, other recaptured turtles had spent up to 27 months in the wild. Most turtles were recaptured a few hundred metres to 14 km away from their point of release, except for one head-started turtle that was recaptured 2,315 km away from the release site after 10 months. In 1967–1971, eighty-nine green turtles were head-started in Costa Rica and released after approximately two years on the north and south coasts of Bermuda. In addition, 19 wild-born immature green turtles caught accidentally by local fisherman were tagged and released as part of the same programme.

A replicated study in 1974–1975 in pelagic waters on the Torres Strait, northern Australia (2) found that a small number of released head-started juvenile green *Chelonia mydas* and hawksbill *Eretmochelys imbricata* turtles were recaptured in the first year after being released. In total, 12 of 1,082 head-started green turtles and 2 of 53 head-started hawksbill turtles were recaptured 12–400 days after being released. All green turtles recaptured were released from one island, two had been released in April–June and 10 had been released in August–October. Turtles had travelled 70–570 km from their point of release. Green and hawksbill turtles were collected as eggs and hatched in captivity or as newly emerged hatchlings. Turtles were kept in captivity and were at least 1–2 years old prior to release in March–October 1974 from different islands (1,082 green turtles released in four cohorts from three islands and 53 hawksbill turtles released in one cohort from one island). Turtles were tagged prior to release. Details of monitoring were not provided.

A replicated study in 1994–1996 in offshore waters in the Caribbean Sea near the islands of Curaçao and Klein Curaçao (3) found that released head-started loggerhead turtles *Caretta caretta* swimming speeds and rest frequency were similar between 1 and 2.5-year-old released turtles, but older released turtles dived more often. Swimming speed and rest frequency was similar between 1–1.5-year-old released head-started loggerhead turtles (speed: 0.4–0.7 m/second; rest frequency: 0–0.8 rests/hour) and 2.5-year-old released head-started turtles (speed: 0.3–0.9 m/second; rest frequency: 0–1.0 rests/hour). Younger turtles dived less frequently than older turtles (1–1.5-year-old: 0–2 dives/hour; 2.5-year-old: 0–4 dives/hour). In August 1993, loggerhead turtle hatchlings from a single nest were collected and reared in captivity in an aquarium for up to 2.5 years. In 1994 (13 individuals, 1–1.5-years old) and 1995–1996 (10 individuals, 2.5-years old) turtles were released onto one of four beaches and allowed to crawl to sea. Turtles were radio tagged and their swimming behaviour was observed from a boat. Turtles were tracked for 45–243 minutes post release (19 individuals).

A controlled study in 1997–2006 in nearshore waters in the Gulf of Mexico, Mexico and USA (4) found that some released head-started female Kemp's ridley turtles *Lepidochelys kempii* survived at least 11 years, nested in the wild and showed similar movement patterns to wild turtles. Eleven female head-started Kemp's ridley turtles were found to have survived 11–19 years in the wild and bred. The authors reported that post-nesting movements and habitat use of the head-started turtles and wild female turtles were similar (data and details of statistical analysis not provided, see original paper for details). Twenty-eight female Kemp's ridley turtles were radio tagged after nesting between 1997 and 2006. Three–six turtles were monitored each year for 9–841 days (5–563 location points/individual). Eleven turtles were released, head-started individuals (two were originally imprinted on Mexico beaches, 9 on Padre Island) and 17 turtles were wild. Head started individuals were reared in captivity for 9–11 months (10 individuals) or 3 years (1 individual) prior to release.

A controlled study in 2005–2006 off the coast of an island in southwestern Japan (5) found that released head-started hawksbill turtles *Eretmochelys imricata* were tracked for several days after release. Four head-started turtles were tracked for 4–9 days, and a fifth turtle was tracked intermittently for 10 months. An additional five wild-caught turtles (held in captivity for 4 months) were tracked for 2–8 days, and two were recaptured 182–199 days after release. Head-started turtles either moved in random directions (four turtles) or stayed at the release site (one turtle), and wild-caught turtles tended to return to their original points of capture. Turtle eggs were collected from a nesting beach on the island and artificially incubated (29°C; >90% humidity), and hatchlings were reared for 2.5 years. An additional five wild turtles were captured and held in captivity for four months. All turtles were fitted with radio transmitters and released in April 2005 following 1 h sea-acclimation in an enclosure net (4 × 4 × 5 m). Turtles were tracked using 12 fixed receivers deployed on the ocean floor (18 m deep).

A replicated, before-and-after study in 1978–2014 on sandy beaches in Texas, USA (6) found that some released female head-started kemp's ridley turtles *Lepidochelys kempii*, that were 'imprinted' by allowing them to crawl to the sea before bringing them in to captivity, returned to nest on or near to the beach that they had been imprinted on at least once. In 37 years, 125 of 916 (14%) nests were laid by 53 different head-started female Kemp's ridley turtles on the beaches where they were imprinted (turtle ages: 10–26 years old, from 12 release cohorts). The first head-started kemp's ridley turtle nests were documented 10–12 years after the nesting females were released and 19 years after the head-start programme began. Over the 37-year programme, 9,204 hatchlings laid by imprinted head-started turtles were released and Kemp's ridley turtle nest numbers laid on the Texas coastline increased to 119 nests in 2014, from near zero in 1979 (nest numbers fluctuated, see original paper for details). In 1978–2000, approximately 22,507 Kemp's ridley turtle eggs were artificially incubated (1,000–2,000 eggs/year). After emergence, hatchlings were released on one of two beaches and allowed to crawl to the sea ('imprinted'), and collected again for head-starting. In 1979–2001, imprinted, individually-marked, head-started turtles were released after 7–33 months in captivity (~23,853 imprinted head-starters released in multiple locations). Head-started nesting females were surveyed in 1986–2014 by beach patrols and satellite tracking. Eggs from all known turtle nests in the USA were collected for artificial incubation until 2014 (1,667 total nests; nesting turtles examined in 916 nests).

A replicated study in Thailand (7) found that captive-reared green turtles *Chelonia mydas* all survived for an eight-week period, and that two of five measures of growth were affected by water depth but not feed type. Survival over an eight-week period was 100%. Final body weight and growth rate were higher in tanks with shallower water (15 cm depth: body weight 107–110 g, growth 2.7–2.8% body weight/day; 30 cm depth: body weight 98–106 g, growth 2.5–2.7% body weight/day), but were not affected by feed type. Body condition, and shell size were similar across all treatments (see paper for details). One hundred and twenty turtle eggs were collected from a single female as part of a head-starting programme, and 103 hatchlings were reared in fibreglass tanks (1.5 m x 0.8 m x 0.8 m) in 15 cm of sea water for 20 days. Turtles were then moved in groups of five to tanks (0.6 x 30 m) with 15 or 30 cm deep water and provided with floating or sinking food pellets (exact number of individuals/treatment not provided).

(1) Burnett-Herkes J. (1974) Returns of green sea turtles (*Chelonia mydas* Linnaeus) tagged at Bermuda. *Biological Conservation*, 6, 307–308.

(2) Kowarsky J. & Capelle M. (1979) Returns of pond-reared juvenile green turtles tagged and released in Torres strait, Northern Australia. *Biological Conservation*, 15, 207–214.

(3) Nagelkerken I., Pors L.P.J.J. & Hoetjes P. (2003) Swimming behaviour and dispersal patterns of headstarted loggerhead turtles *Caretta caretta. Aquatic Ecology*, 37, 183–190.

(4) Shaver D.J. & Rubio C. (2008) Post-nesting movement of wild and head-started Kemp's ridley sea turtles *Lepidochelys kempii* in the Gulf of Mexico. *Endangered Species Research*, 4, 43–55.

(5) Okuyama J., Shimizu T., Abe O., Yoseda K. & Arai N. (2010) Wild versus head-started hawksbill turtles Eretmochelys imbricata: post-release behavior and feeding adaptions. *Endangered Species Research*, 10, 181–190.

(6) Shaver D.J. & Caillouet Jr C.W. (2015) Reintroduction of Kemp's Ridley (*Lepidochelys kempii*) sea turtle to Padre island national seashore, Texas and its connection to head-starting. *Herpetological Conservation and Biology*, 10, 378–435.

(7) Songnui A., Thongprajukaew K., Kanghae H., Satjarak J. & Kittiwattanawong K. (2017) Water depth and feed pellet type effects on growth and feed utilization in the rearing of green turtle (Chelonia mydas Linnaeus, 1758). *Aquatic Living Resources*, 30, 18.

Tortoises, terrapins, side-necked & softshell turtles

- **Eighteen studies** evaluated the effects of head-starting wild-caught tortoises, terrapins, side-necked and softshell turtles for release. Thirteen studies were in the USA[2,3,5,7,9-13,15-18], two were in Venezuela[5,14] and one was in each of the Galápagos[1], Poland[4] and Madagascar[8].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (18 STUDIES)

- **Abundance (1 study):** One controlled study in Venezuela[14] found that 57% of captured giant sideneck river turtles were head-started individuals.
- **Survival (13 studies):** Two of three studies (including one replicated, controlled study) in the USA[2,16] and Poland[4] found that head-started European pond turtles[4] and desert tortoises[16] had similar survival compared to wild turtles[4] or hatchlings released directly into the wild[16]. The other study[2] found that head-started northern redbelly turtles had higher survival than wild hatchling turtles. This study[2] also found that in the first year of release, larger head-started turtles had higher survival, but in year 2–3 survival was similar for all sizes. Four of 12 studies (including nine replicated studies) in the Galápago[1], the USA[3,7,9-13,15,18], Madagascar[8] and Venezuela[14] reported that 50–100% of head-started individuals survived for three months to 1–5 years after release[1,3,10,12]. Three of the studies[8,13,18] reported that 6–43% of individuals survived for 1–3 years. Two of the studies[9,11] reported that six of six[9], two of 10 and nine of 10[11] radio-tracked individuals survived 3–12 months. Two of the studies[7,15] reported that annual survival was 80–100%[7] or 3–100% in the year following release but 82–100% in subsequent years[15]. The other study[14] reported that some giant sideneck river turtles survived up to 14 years[14]. Two studies[9,12] also reported that survival during the captive phase was 91–100%. One study[1] also found that more tortoises head-started in outdoor seaside pens died than did those from indoor pens. One replicated, controlled study in Venezuela[5] found that survival of Arrau turtles during the captive phase was lower for turtles from relocated nests compared to those from nests that were not moved.
- **Condition (5 studies):** One of two replicated studies in the USA[11,17] found that two-year-old head-started gopher tortoises were larger at their time of release than two-year-old tortoises released in to the wild directly after hatching[11]. The other study[17] found that Agassiz's desert tortoise hatchlings grew more slowly in captivity than tortoises in the wild. Two studies (including one replicated study) in the USA[5,12] found that Alabama red-bellied cooters[5] and wood turtles[12] grew during 12–16 months in captivity, and wood turtles showed no signs of shell malformation[12]. One controlled study in Venezuela[14] found that the size distribution of released head-started giant sideneck river turtles was similar to that of wild turtles when newly released individuals were excluded.

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One study in the USA[10] found that 81% of desert tortoises established home ranges within 13 days of release.

A replicated study in 1965–1972 in a captive rearing facility and on an island in Galápagos, Ecuador (1) found that around two thirds of head-started Galápagos giant tortoise *Geochelone elephantopus* of five subspecies survived captive rearing and that over half of released juvenile *Geochelone elephantopus ephippium* survived at least 8 months in the wild after release. Results were not statistically tested. At least 41% (51 of 124) of head-started hatchlings kept in outdoor seaside pens died within the first 18 months compared to 18% (31 of 172) reared indoors or in a bespoke rearing facility. From two releases of head-started individuals, 20 of 20 (100%) and 25 of 51 (50%) tortoises survived 8–10 month, and 12 of 20 (60%) survived at least 17 months. Authors reported no instances of ill health; that all recaptured tortoises had increased in size and weight; and that individuals from the first release were heavier five and 10 months after release than equivalent captive animals (see paper for details). In 1965–1971, giant tortoise hatchlings were reared in captivity (including some captive-bred and some wild caught hatchlings; see paper for subspecies). The 1965/1966–1967/1968 cohorts (124 hatchlings) were reared in outdoor fenced seaside pens. The 1968/1969 cohort (50 hatchlings) were reared indoors. From 1970 onwards all cohorts were reared in a bespoke facility (122 hatchlings). Twenty tortoises were released in December 1970, and 51 were released in May 1972, and all were monitored for up to 19–17 months.

A controlled study in 1979–1988 in one large pond in Massachusetts, USA (2) found that released head-started northern redbelly turtles *Pseudemys rubriventris* had higher survival than translocated, wild hatchlings, and that larger head-started turtles had higher survival than smaller ones. Annual survival of head-started turtles (36–100% of 12, 13 and 38 turtles released/year) was higher than for translocated hatchlings (0 of 15, 0%). Larger head-started turtles had higher annual survival in the first year following release (<65 mm: 36%; 66–95 mm: 66%; ≥96 mm: 92%), but in year 2–3 after releases survival was similar for all sizes (60–100%). Three of five additional turtles raised in captivity for one year were re-captured 13 years later. Hatchling turtles were collected from a nearby pond and raised in a head-starting facility for around 9–12 months. In 1979–1988, a total of 68 head-started turtles were released (5 in 1979, 62 in 1985–1988), and in 1982, fifteen wild hatchlings were translocated immediately after capture. In 1985–1992, turtles were trapped annually over a 4–6-week period from May–June or August–September using basking traps and fyke nets.

A study in 1994–2001 in an altered waterway in an urban setting in California, USA (3) found that some released head-started western pond turtles *Actinemys marmorata* survived for 1–5 years after release. Hatching success of artificially incubated eggs from wild-caught females was 53%. Twenty-one of 33 (64%) head-started turtles were recaptured at least once, 1–5 years following release. In 1994–1998, some wild-caught, gravid females were hormonally induced, and eggs were collected and incubated in moist vermiculite. Hatchlings were raised for six months (4 individuals) or two years (27 individuals) and then released. Turtles were captured by hand, dip net, basking net and baited traps, as well as collecting turtles in 1997 and 1998 from the drained wetland while maintenance was occurring.

A replicated, controlled study in 1997–2002 in wetlands in Borowiec Nature Reserve, central Poland (4) found that released head-started European pond turtles *Emys orbicularis* had similar survival compared to wild turtles. Annual survival was similar for head-started (1-year-olds: 21–35%; 2-year-olds: 43–70%; 3-year-olds: 44%) and wild-caught turtles (Hatchlings: 5–25%; 1-year-olds: 64–46% and 0–100%; 2-year-olds: 100%). In 1997–2000, nesting females were monitored, and during September, hatchlings and eggs from 3–13 clutches/year were removed for head-starting. They were raised in groups of 10–15 (40 x 50 cm aquarium; water temperature 20°C) and fed live insect and earthworm prey. Head-started individuals were marked released at one year of age (69 in 1998; 34 in 1999; 20 in 2000). Turtles were monitored by capturing with a dip net and baited traps from April–August and any wild turtles were marked.

A replicated, controlled study in 2003 on one river in Venezuela (5) found that first-year mortality of Arrau turtles *Podocnemis expansa* during head-starting in captivity was higher for turtles from relocated nests compared to those from naturally incubated nests. First-year mortality was higher for turtles from relocated nests compared to natural nests (relocated: 13 of 108, 12%; natural: 1 of 112, <1%). Turtles from relocated nests had more physical abnormalities than naturally incubated turtles in two locations on the shell (relocated: 74%, 77%; natural: 19%, 33%), but a similar number at a third location on the shell (relocated: 4%; natural: 5%). There was no significant difference in hatching success between relocated and natural nests (54–98%). In February 2003, six nests were excavated and reburied 1.5 km further up the riverbank. In April 2003, a total of 230 hatchlings from the relocated nests and four naturally incubated nests (up to 28 turtles/nest) were collected. Turtles were head-started in a holding tank for up to one year and fed with high-protein fish meal before being released at their beach of origin.

A study in 2004–2005 in a river delta site and a captive setting in Alabama, USA (6) found that six head-started Alabama red-bellied cooters *Pseudemys alabamensis* grew and survived 16 months in captivity before they were released. Six wild hatchlings brought into captivity increased their weight from 15 g to 311 g and their size (carapace length) from 38 mm to 126 mm over 16 months. In 2004–2005, six hatchlings were rescued from a causeway near some nesting sites and were brought into captivity. Hatchlings were raised in a 55 gallon aquarium for 16 months and released near their point of capture.

A replicated study in 1999–2004 in a wetland site with a lake and ponds in Washington, USA (7) found that released head-started western pond turtles *Actinemys marmorata* had high survival over 1–4 years following their release. Annual survival of head-started was estimated at 80–100%, with 0–5 turtles found dead each year (of 16–46 turtles monitored/year). Hatchling turtles were collected from nests in September–October 1999–2002 and were head-started in local zoos for 10–11 months before release. Turtles that had not reached 50 g were held for an additional year. Head-started turtles were marked and released in 2000 (40 turtles), 2001 (38), 2002 (59) and 2003 (51). A subset of turtles (16–20 turtles/year, 68 in total) were radio tagged and relocated 1–3 times/per week, or once/week during winter.

A replicated study in 1998–2011 at a lakeside in Ankarafantsika, Madagascar (8) found that around a third of released head-started Madagascar big-headed turtles *Erymnochelys madagascariensis* survived on year in the wild and less than half of the first-year survivors were still alive after three years. Of head-started Madagascar big-headed turtles released in 2004, a total of 47 of 158 turtles (30%) survived one year, 32 of 158 turtles (20%) survived two years and 20 of 158 turtles (13%) survived three years in the wild. From 1998–2011, two hatchling Madagascar big-headed turtles were taken from each wild nest laid in a healthy population at Lake Antsilomba and raised in captivity for 1–7 years (280 nests and 410 hatchlings collected). Head-started turtles were released in 2004 (158 individuals, 3–5 years old) and 2009 (180 individuals, 1–7 years old) at Lake Ankomakoma. The population at Lake Ankomakoma was monitored twice a year after releases. Only results from the 2004 release were reported.

A replicated study in 2006–2011 in forested wetlands in eastern Massachusetts, USA (9) found that most head-started Blanding's turtles *Emydoidea blandingii* survived in captivity, and after being released, some survived in the wild for at least nine months. Survivorship of head-started hatchling Blanding's turtles in captivity was 91–100% (2006: 0 of 7 hatchlings died; 2007: 0 of 22 hatchlings died; 2008: 3 of 31 hatchlings died; 2009: 3 of 47 hatchlings died; 2010: 3 of 54 hatchlings died). One head-started turtle from 2008 lost weight when recaptured five months after release (weight at release: 164 g; five months later: 143 g) but survived a year in the wild before dying. Five head-started turtles from 2009 survived at least nine months in the wild although one of the five turtles died before the end of the first winter. In August 2006–2010, wild Blanding's turtle hatchlings were collected for head-starting and release at a wildlife refuge (reserve size: 880 ha; 2006–2010: 161 hatchlings taken from 59 nests). Head-started turtles were maintained in aquariums/plastic containers and fed regularly (see original paper for details). Most head-started turtles were released seven months after hatching (late May, except a group in 2006 which were kept in captivity for a year). One head-started hatchling in 2008 and five in 2009 were released with radio transmitters and tracked for up to one year.

A study in 2001 in desert scrub in California, USA (10) found that over half of released head-started juvenile desert tortoises *Gopherus agassizii* survived at least three months in the wild. After 90 days in the wild, nine of 16 (56%) released head-started juvenile tortoises were still alive. The seven dead tortoises died from predation. The authors reported that 13 days after release, 13 of 16 tortoises (81%) had settled into home ranges, and that 54 days after release, tortoises were an average of 174 m from their release location. From 1989, wild-caught hatchling tortoises were reared in predator-proof enclosures. In March 2001, sixteen juvenile tortoises (8–9 years old) were released 500 m from their rearing pen and radio-tracked for 91 days during daylight hours. Tortoises were located 16 times after release.

A replicated study in 2010–2013 in two pine forest sites in Mississippi, USA (11) found that almost two-thirds of released head-started gopher tortoises *Gopherus polyphemus* survived at least three months in the wild and while in captivity grew bigger than captive-born gopher tortoises of the same age that were released immediately after hatching. In one site, nine of 10 radio-tracked head-started gopher tortoises survived at least one year after release. At a second site, two of 10 radio-tracked head-started gopher tortoises survived at least three months (of the remaining tortoises, seven were predated and the fate of one was unknown). Two-year-old head-started hatchlings were longer and weighed more (carapace length: 97 mm, weight: 204 g) at time of release than two-year-old tortoises that were released into the wild immediately after hatching (carapace length: 62 mm, weight: 53 g). In 2010, ninety-three gopher tortoise eggs were collected from two locations in the wild in May–June and relocated for artificial incubation. Thirty-one hatchlings were head-started for two years in individual indoor containers (see paper for details), and 20 hatchlings were released immediately after hatching. From June–July 2012, head-started hatchlings were housed in individual outdoor enclosures (2 x 2 m). They were released in August 2012 into two sites (after five days in release pens), and 10 tortoises/site were radio tracked three times/week until the end of the radio transmitters lifespan. Head-started tortoise size/weights were compared to the 20 two-year-old tortoises released as hatchlings at one of the release sites.

A replicated study in 1994–2001 in an indoor enclosure in New York State, USA (12) found that all head-started hatchling wood turtles *Glyptemys insculpta* survived at least one year in captivity prior to release and at least two years in the wild after release. In total, 11 of 11 head-started wood turtles survived one year in captivity and four of four survived two years in captivity. All turtles grew rapidly and uniformly while in captivity, with no signs of shell malformation. All 10 head-started juvenile wood turtles released into holding pens prior to their main release survived at least two years in the wild. Nine of 10 turtles were subsequently recaptured as sub-adults or adults (years were not provided) and one turtle was re-captured eight years after being released. The authors reported that no differences in movement behaviour after release were observed between one and two-year-old turtles. Wood turtle juveniles that were hatched from wild-collected eggs in 1994 (eight hatchlings), 1998 (3 hatchlings) and 1999 (4 hatchlings) and head-started indoors were released into holding pens in the wild after one or two years in captivity (six individuals were released as 1-year-olds and four as 2-year-olds). Turtles were placed in outdoor predator-proof plywood and cloth enclosures (size: 122 x 183 cm or 122 cm^2) for four–six weeks before their main release. Turtles were radio tracked for 1–3 seasons and monitored on an ad hoc basis from then on (see original paper for details).

A replicated study in 2003–2008 in a desert scrub site in California, USA (13) found that very few released head-started Agassiz's desert tortoises *Gopherus agassizii* survived for a year after release. Three of 47 tortoises (6%) survived for a year after release, with the rest dying from predation or other causes. Enclosures were constructed in a natural habitat setting using fencing and mesh netting, with six being irrigated and nine receiving only natural rain. Irrigated pens received 25–38 mm of water through a sprinkler system once in late winter and twice in spring. From 2003, wild, adult females (number not given) were brought into the pens to lay eggs before being re-released. Yearling tortoises (47 individuals) were released in 2004–2007, with 31 coming from irrigated enclosures and 16 from natural enclosures. Thirty-one were release close to the enclosures and 16 were released 1 km away. All were fitted with radio transmitters and located every two weeks during active time periods and monthly during winter.

A controlled study in 2008 in a river basin in middle Orinoco, Venezuela (14) found that released head-started giant sideneck river turtles *Podocnemis expansa* survived in the wild for up to 14 years. Eighteen years after the start of a conservation programme, 99 of 174 (57%) giant sideneck turtles caught during monitoring were head-started turtles that had survived in the wild for up to 14 years. Excluding newly released head-started turtles, the size distributions of head-started and wild turtles were similar (see original paper for details). The authors reported that head-started turtles were yet to reach a size comparable to mature wild turtles. In 1992, a programme to head-start and release giant sideneck river turtles after one year in captivity began, with around 350,000 turtles released in total. In April–June 2008, turtle surveys were carried out along a 50 km stretch of river by pulling a trawl net (5 cm mesh) between two boats, or between one boat and people on land. All caught turtles were classified as head-started or wild, measured, sexed, and individually marked before being re-released.

A replicated, site comparison study in 2002–2010 in two open mixed pine forests in South Carolina and Georgia, USA (15) found that survival rates of released head-started gopher tortoises *Gopherus polyphemus* were extremely variable in the first year following release, but consistently improved in the second to fourth years. Results were not statistically tested. Across three groups of released head-started tortoises, 3–100% survived the first year in the wild (2002 group: 17 of 32, 53% individuals survived; 2006: 7 of 7, 100%; 2007: 1 of 32, 3%). Survival rates improved overall in the subsequent three years after release (2002 group: 82–93%; 2006: 100%; 2007: 100%). In total, 97–100% of head-started gopher tortoise hatchlings survived the captive rearing period. In 2002, 2006 and 2007, head-started gopher tortoise hatchlings were released into two sites (2002: 32 hatchlings in an 80,000 ha forest reserve; 2006 and 2007: 7–32 hatchlings, released on a 5,670 ha island). Hatchlings were head-started in climate-controlled indoor enclosures from the autumn after hatching until the following spring, when they were released into enclosures (one/site) with artificial burrows. Enclosures were removed approximately six months later. Hatchlings were monitored by live trapping for two weeks in September–October 2002–2006 (forest reserve site) and 2006–2010 (island site).

A replicated, before-and-after study in 2015–2016 in desert scrubland in California, USA (16) found that survival during head-starting of desert tortoises *Gopherus agassizii* was similar compared to survival of tortoises that were released directly into the wild after hatching over six months. Survival rates of head-started and directly released juvenile desert tortoises were similar after six months (head-started indoors: 29 of 30, 97% survived; head-started outdoors: 20 of 20, 100% survived; directly released: 15 of 20, 75% survived). In August–September 2015, eggs from 25 wild female tortoises were collected and incubated outdoors in artificial burrows. In September 2015, seventy hatchlings (21–46 days old) were moved to either an indoor enclosure (30 hatchlings), outdoor enclosure (20 hatchlings; 30 x 30 m, semi-natural enclosure) or were released directly into the wild (20 hatchlings). Food was provided in indoor and outdoor enclosures (see paper for husbandry details). Directly released hatchlings were released in a 0.7 km^2 unfenced area and monitored using radio telemetry once or twice/week until March 2016.

A replicated study in 2003–2012 in a desert region of California, USA (17) found that some head-started Agassiz's desert tortoises *Gopherus agassizii* survived the head-starting process, but growth was slower than in two wild populations. Growth of head-started tortoises was slower (4 mm/year) than in two wild populations (9–10 mm/year). After seven years, the captive facility contained 261 tortoises (1,718 tortoises/ha; range: 789–2,758 tortoise/ha), and after the following two years there were 142 tortoises (900 tortoises/ha; range: 0–2,049 tortoise/ha). In 2003–2010 (months not specified), around 24 adult females were brought in to one of nine enclosures to lay eggs before being returned to the wild. No further females were brought into the enclosures in 2011–2012, and no captive-born individuals are reported to have bred. The nine enclosures ranged from 0.01–0.03 ha in size and were covered with mesh to exclude avian predators and reinforced with exclusionary fencing at the base. Counts of tortoises in each enclosure in 2003–2012 (months not specified) were used to calculate density of tortoises/ha. Tortoises raised in the enclosures were intended for release into the wild.

A replicated, randomized study in 2013–2016 in two sites in a mixed pine forest in Georgia, USA (18) found that less than half of released head-started gopher tortoises *Gopherus polyphemus* that were held in pens prior to release survived one year after release. In two consecutive years, less than half of head-started gopher tortoises survived for one year after release (year one: 5 of 12, 42% tortoises survived; year two: 13 of 30 43%). The authors reported that the primary cause of mortality was predation by mammals or fire ants and that 71% of all mortalities occurred in the first 30 days after release. In 2013–2014, wild tortoise eggs were collected and incubated in captivity, and hatchlings were head-started for 8–9 months indoors. In total, 145 tortoises (July 2014: 12 individuals; July 2015: 133 individuals) were released into two sites in a protected area (3,127 ha). Survival estimates were based on a subset of tortoises (2014: 11 individuals; 2015: 30 individuals) that were radio tracked for one year after release. Tortoises were placed in predator-resistant, enclosed holding pens (5–6 tortoises/pen, random groupings) for 4–47 days prior to release. Each pen contained 5–10 artificial burrows (30–40 cm deep). Release sites were sprayed with insecticide (AMDRO®) to remove fire ants up to a 3 m perimeter around the edge of release pens.

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Snakes & lizards

- **Nine studies** evaluated the effects of head-starting wild-caught snakes and lizards for release. Five studies were in the USA[1-3,6,8], two were in Puerto Rico[4,5] and one was in each of the Cayman Islands[7] and Jamaica[9].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (9 STUDIES)

- **Abundance (2 studies):** Two studies (including one before-and-after and one replicated study) in Jamaica[9] and the Cayman Islands[7] reported that the number of Jamaican iguanas found in the wild was higher after 23 years of head-starting and releasing compared to at the start of the programme[9] and that there was a stable population of blue iguanas over four years during ongoing releases of head-started individuals[7].

- **Reproductive success (4 studies):** Four studies (including two replicated studies) in Jamaica[9], Puerto Rico[4,5] and the USA[1] reported successful reproduction following release of head-started Jamaican iguanas[9] (but not for 16 years) and Mona Island iguanas[4,5], and that timber rattlesnakes copulated or participated in pre-copulatory behaviour[1]. One study[5] also reported that 88–90% of Mona Island iguana eggs hatched successfully[5].

- **Survival (8 studies):** Two of three controlled studies (including one replicated, randomized study) in the USA[3,6,8] found that head-started plains gartersnakes[3] and common water snakes[8] were recaptured a similar number of times[3] or had similar survival[8] compared to resident snakes. The other study[6] found that head-started northern water snakes had lower survival following release than resident snakes. One study[3] also found that 76% of snakes survived the captive phase of head-starting. Three studies (including two replicated studies) in the USA[1] and Puerto Rico[4,5] reported that 22–40% of timber rattlesnakes[1] or Mona Island iguanas[4,5] survived for monitoring periods of eight months to six years. One replicated study in the USA[2] found that head-started eastern massasaugas released in summer had higher survival than snakes released in autumn. One before-and-after study in Jamaica[9] reported that 16% of Jamaican iguanas died during the captive phase of head-starting.
- **Condition (5 studies):** Two of three controlled studies (including one replicated, randomized study) in the USA[3,6,8] found that head-started northern water snakes[6] and common water snakes[8] grew more slowly than resident snakes. The other study[3] found that head-started plains gartersnakes had similar growth rates to resident snakes. One study[8] also found that head-started common water snakes had similar body condition to resident snakes. One controlled study in Puerto Rico[5] found that body condition of head-started Mona Island iguanas was higher than wild iguanas before release, but similar at their first recapture after release. One replicated study in the USA[2] found that more head-started eastern massasaugas released in summer gained weight before hibernation than snakes released in autumn.

BEHAVIOUR (3 STUDIES)

- **Behaviour change (3 studies):** One of three studies (including one replicated, randomized, controlled study) in the USA[2,6,8] found that head-started common water snakes showed similar behaviour to residents across a range of behaviour measures[8]. One of the studies[6] found that head-started northern water snakes had smaller home ranges and showed less surface activity than resident snakes. The other study[2] found that head-started eastern massasaugas released in summer had larger home ranges than snakes released in autumn.

A replicated study in 1993–1999 in a hardwood forest in eastern Texas, USA (1) found that some released head-started timber rattlesnakes *Crotalus horridus* survived for at least 2–6 years following release. Eight of nine released snakes survived for one year following release and at least two survived for six years. The status of a further three snakes was unknown after two years. Three of nine head-started rattlesnakes were observed mating or participating in pre-mating behaviour five years after release. Nine young snakes (8 from a single adult female tracked near the eventual release site) were captured and housed in individual cages for six months (1 snake), 12 months (4 snakes) or 18 months (4 snakes). Snakes were released in March 1994 (1 snake), August 1995 (4 snakes) or February 1996 (4 snakes). All snakes were surgically implanted with transmitters and located weekly in March–November for 4–6 years.

A replicated study in 1999–2001 in two sites of mixed wetland and scrub oak in Wisconsin, USA (2) found that head-started eastern massasaugas *Sistrurus* *catenatus catenatus* released in summer had lower mortality rates, larger home ranges and gained more mass compared to snakes released in autumn. Summer released snakes had lower mortality (7 of 15, 47% snakes died during hibernation) than autumn release snakes (14 of 15, 93% died either before, during or immediately after hibernation). Summer released snakes had larger home ranges (12 ha) than autumn release snakes (1 ha), and 65% (11 of 17) of summer released snakes gained weight prior to hibernation compared to 0% (0 of 15) of autumn-released snakes. Pregnant female snakes from three locations in Wisconsin USA were captured and 50% of each brood was retained for head-starting. Thirty-two head-started snakes were released with radio transmitters either in September 1999 (15 snakes, 1–3 years old) or July 2000 (16 snakes, two years old) and located daily after release until hibernation. All surviving snakes were re-captured in April 2001 and placed back in captivity.

A replicated, controlled study in 1995–2001 on an urban river bank with a mix of mown lawns and riparian vegetation in Illinois, USA (3) found high survival during head-starting of plains gartersnakes *Thamnophis radix*, and that post-release survival was comparable to wild-caught snakes. Overall survival during head-starting was 76% (217 of 286 snakes). The number of snakes recaptured one or more years after release was similar for head-started (5% of 142 snakes and 26% of 53 snakes) and wild-caught snakes (20% of 80) (result not statistically tested). Growth rate was similar for head-started and wild-caught snakes (data reported as statistical model result). Three head-started females were gravid when recaptured (23–24 months old). In 1995–2001, gravid females were captured (number not given) and maintained in captivity until giving birth. Snakes born in 1995 and 1996 (53 snakes) were head-started for 327–335 days, while those born in 1999 (142 snakes) were head-started for 253–260 days. Recapture effort varied between months and years, but most snakes were recaptured by hand in April–June 1998–2001.

A replicated study in 1999–2004 in a subtropical dry forest site on Mona Island, Puerto Rico (4) found that some released head-started Mona Island iguanas *Cyclura cornuta stejnegeri* survived in the wild and two females were observed breeding. Forty percent (4 of 10) of iguanas survived >1 year in the wild, and at least 30% (3 of 10) survived >2 years. At least 50% of the females (2 of 4) bred in 2004. Four of five (80%) iguanas released at their point of capture survived at least 96 days and two of five (40%) released at a new site survived at least 33 days. Hatchlings were collected from the wild in November 1999 and reared until they reached a target size (snout-vent-length: 25 cm; mass: >950 g). Iguanas were implanted with radio transmitters (12 g) and marked with PIT tags and coloured beads on the crest. In April–August 2002, five individuals were released at their point of capture, and five at a new site. All iguanas were monitored daily until radio transmitters failed, and monitored by active searching thereafter.

A controlled study in 1999–2006 in a subtropical dry forest site on Mona Island, Puerto Rico (5) found that some released head-started Mona Island iguanas *Cyclura cornuta stejnegeri* survived up to five years in the wild and at least two reproduced successfully. Twenty-five of 62 (40%) head-started iguanas (15 females, 10 males) were re-captured between 8–61 months following release. Two head-started females produced clutches of 8 and 11 eggs each, and 86–91% of eggs hatched successfully. Body condition of head-started iguanas was higher before release than wild iguanas but was similar at their first capture post-release (data reported as condition index). Sixty-two hatchlings were collected from the wild by digging up nests in October 1999 (8 nests) and 2000 (6 nests), and transported to fenced enclosures. Hatchlings were marked with PIT tags and weighed and measured every 4 months while in captivity. Iguanas were marked and released at their nesting site in April 2002 and October 2003 after reaching at least 620 g and 225 Snout to vent length (about 3 years of age). Seven mid-sized and 31 adult wild iguanas were also captured and measured. Intensive trapping was conducted during 2–3 months in 2003–2006.

A controlled study in 2008–2009 in a site of mixed hardwood forest and scrub patches in Indiana, USA (6) found that head-started northern water snakes *Nerodia sipedon sipedon* had lower survival than resident snakes, as well as lower movement and growth. Following release, 8 of 12 head-started snakes survived until hibernation (67% survival over 5 months), but none survived one year, whereas seven of 12 resident snakes survived until hibernation (58% survival) and four survived to the end of the year (33% survival). Head-started snakes had smaller home ranges (head-started: 2 ha; resident: 5 ha) and grew less than resident snakes (head-started: 0.03 cm/day & 0.05 g/day; resident: 0.07 cm/day & 0.80 g/day). Head-started snakes also showed less surface activity than residents (reported as activity index). In July 2007, seven pregnant snakes were captured and gave birth in captivity before being returned to their capture site. Sixty newborn snakes (30 females, 30 males) were housed in small plastic boxes (20 x 65 x 13 cm) for 11 months and 12 snakes (9 females, 3 males) were chosen for release. Twelve resident snakes (matched in terms of size and sex) were captured in May 2008. All snakes were implanted with radio transmitters and were located once/week from May–September, every two weeks from October–November and March–April, and monthly from December–February.

A replicated study in 2004–2013 in a dry shrubland site in a reserve on Grand Cayman, Cayman Islands (7) found that releasing head-started blue iguanas *Cyclura lewisi* resulted in a stable population over four years, but that the population size remained lower than the number of iguanas released. Six years after the first head-started iguanas were released, but while releases were ongoing, 46 and 42 iguanas were re-sighted in 2010 and 2013 respectively, and densities were estimated at 5–6 iguanas/ha. Authors reported that released iguanas were also sighted outside of the study area. In 2004–2009, a total of 307 head-started iguanas were released, and in 2010–2012, a further 98 iguanas were released. All iguanas were tagged at time of release with unique coloured glass bead combinations (as piercing on the neck crest) and PIT tags. In the first three weeks of March 2010 and 2013, iguana surveys were conducted twice a day by teams of two observers on 12 transects of unequal length (range 323–432 m). All offspring that were captured were also tagged.

A replicated, randomized, controlled study in 2007–2010 in mixed wetland, shrubland and hardwood forest in Indiana, USA (8) found that common water snakes *Nerodia sipedon sipedon* that were released following two methods of head-starting had similar survival rates and showed similar behaviour compared to resident snakes, but grew more slowly. Annual survival following release was similar for head-started snakes (basic conditions: 64%, semi-natural conditions: 50%) and resident snakes (46%). A range of behaviour and activity measures, including post-release movement, habitat use, and hibernation date were also similar between head-started and resident snakes (see paper for details). Head-started snakes grew more slowly than resident snakes (head-started: 0–0.05 cm/day; resident: 0–0.11 cm/day), but body condition remained similar between all groups (data presented as statistical model result). In July 2007, seven pregnant female snakes were captured and brought into captivity. Sixty offspring were raised for 18 months in individual plastic tubs (20 x 64 x 13 cm) containing a water bowl and hide. In February 2009, snakes were divided in to two groups and raised for a further four months in either basic conditions (remaining in the plastic tub) or semi-natural conditions (see paper for details). After 22 months in captivity, head-started snakes (basic conditions: 12 snakes; semi-natural conditions: 10 snakes) and an additional 15 resident wild snakes were released, and radio-tracked 1–4 times/month throughout the year. Of the resident snakes, eight were tracked in 2008–2009, three in 2009–2010, and four during both seasons.

A before-and-after study in 1991–2015 in old-growth tropical dry forest in Jamaica (9) found that after releasing head-started Jamaican iguanas *Cyclura collei* (along with associated actions), the number of nesting female and hatchling iguanas increased over 23 years. Results were not statistically tested. After 23 years of head-starting and releasing Jamaican iguanas, 321 iguana hatchlings and 63 nesting female iguanas were counted compared to 31 hatchlings and 9 nesting females at the start of the programme. The first new wild-born female iguana joined the breeding population after 16 years. The authors reported that health of head-started individuals was generally good but that 16% died or were lost prior to being released. In 1991–2015, Jamaican iguana eggs and hatchlings were collected from the wild and head-started in a zoo. Head-started individuals were released in 1996 (278 total iguanas released, usually 6–8 years old or 1–2 kg). In 1997–2014, non-native mammalian predators (mongoose *Herpestes javanicus*, cats *Felis catus*, dogs *Canis lupus familiaris* and feral pigs *Sus scrofa*) were removed using baited cage traps, snares and leg-hold traps (around 1,500 predators in 350,000 trap days over 17 years using 20–300 cage traps). In 2011–2012, an artificial nesting site was constructed 40 m south of the main nesting area. During the nesting season in 1991–2015, nests were checked daily and adult female iguanas were monitored by live trapping, observation and camera traps.

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(4) García M., Pérez-Buitrago N., Álvarez A. & Tolson P. (2007) Survival, dispersal and reproduction of headstarted Mona Island iguanas, *Cyclura cornuta stejnegeri*. *Applied Herpetology*, 4, 357–363.

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Crocodilians

- **Seven studies** evaluated the effects of head-starting wild-caught crocodilians for release. Two studies were in each of the Philippines[4,6] and Nepal[5,7] and one study was in each of Zimbabwe[1], Venezuela[2] and Argentina[3].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (7 STUDIES)

- **Abundance (2 studies):** Two studies (including one replicated study) in the Philippines[6] and Nepal[7] reported that following releases of head-started crocodiles[6] or gharials[7], wild populations increased in size over 8–9 years.
- **Reproductive success (2 studies):** One replicated study in Argentina[3] reported that released head-started broad-snouted caimans had similar clutch sizes and hatching success compared to non-head-started caiman. One replicated study in Nepal[7] reported successful reproduction in all four rivers where head-started gharials were released.
- **Survival (5 studies):** Three studies (including one replicated, controlled study) in Venezuela[2], the Philippines[4] and Nepal[5] reported that 88% of head-started Orinoco crocodiles survived 8–12 months[2] and 53% of Philippine crocodiles[4] or gharials[5] survived for one year[4] following release. One study[4] also found that survival of Philippine crocodile hatchlings during the captive phase of head-starting was higher than for non-head-started hatchlings in the wild. One replicated study in Argentina[3] reported that at least five released head-started broad-snouted caimans survived 9–10 years. One replicated study in Zimbabwe[1] found that 38% of released head-started Nile crocodiles were recaptured at least once over four years. This study[1] also found that hatching success of Nile crocodile eggs in the head-start programme was 74%, and that survival of hatchlings during the captive phase was lowest during the first year.
- **Condition (1 studies):** One study in Venezuela[2] found that released head-started Orinoco crocodiles grew at a similar rate to resident juvenile crocodiles.

BEHAVIOUR (0 STUDIES)

A replicated study in 1967–1974 in three sites along the Zambezi River in Zimbabwe (1) found that some released head-started Nile crocodiles *Crocodylus niloticus* survived at least six months in the wild, and that mortality during head-starting was highest during the first year. Over seven years, hatching success of Nile crocodile eggs in a head-start programme was 74% (16,697 of 22, 697 eggs hatched). In one site, hatchling mortality from six annual cohorts was 8–52% in the first year, 1–14% in the second year and 0–4% in the third year (see original paper for further details). Twenty of 53 (38%) released head-started crocodiles were caught at least once in four years following release (see original paper for details). In 1967–1973, Nile crocodile eggs were collected from the wild, and hatchlings were head-started at three rearing stations (at Kariba, Binga and Victoria Falls) as part of a crocodile farming initiative (128–2,475 eggs collected/station/year). Eggs were artificially incubated in captivity (no details are provided). An annual quota set by the government required 5% of three-year-old crocodiles were returned to the wild. In total, 355 head-started crocodiles were returned to the wild by the end of 1973, of which 53 released into one site were monitored by twice-yearly recapture surveys in 1970–1974.

A study in 1991–1992 in a river near San Jose, southwestern Venezuela (2) found that after releasing head-started Orinoco crocodiles *Crocodylus intermedius*, some survived at least a year. Seven crocodiles survived for at least 235–352 days, and one was killed accidentally two weeks following release. Crocodiles moved an average of 4–5 km/month (maximum distance 12 km). The average growth rate of released crocodiles (4 of 8 released individuals) was 0.1 cm/day, which was comparable to some smaller, wild-caught juveniles (0.1 cm/day) (result not statistically tested). In 1987, eggs were collected from along the river and hatched in captivity. Eight male crocodiles were head-started (length range from 115–139 cm) and released in March or April 1991. Crocodiles were fitted with radio transmitters and located every 1–2 days from April 1991 to March 1992.

A replicated study in 2001–2002 in Santa Fe province, Argentina (3) found that some released head-started female broad-snouted caiman *Caiman latirostris* survived at least 9–10 years and bred in the wild. Seven released head-started female caiman (Five 9-year-olds and two 10-year-olds) nested within 1 km of their release sites. Clutch size and hatching success of wild-collected caiman nests was similar to wild nests left in situ (wild-collected clutch size: 26–41 eggs/nest and hatching success: 43–100%; wild in situ nests: no data provided). Since 1990, a head-starting programme collected caiman eggs from wild nests in December–January and artificially incubated the eggs. Hatchlings were head-started for up to nine months and then released back into the wild at the collection site (see original paper for details). In austral summer 2001–2002, seven head-started female caiman were captured while guarding their nests. Eggs were collected from the nests and artificially incubated. Head-starter clutch size and hatching success data were compared with nests (clutch size comparison was with 31 nests; hatching success comparison was with 11 nests).

A replicated, controlled study in 2005–2009 in a captive facility and ponds, creeks and rivers in northern Luzon, Philippines (4; same experimental set-up as 6) found that most head-started Philippine crocodiles *Crocodylus mindorensis* survived rearing in captivity and at least half survived their first year in the wild. After one year in captivity, head-started Philippine crocodile hatchling survival was 72% (63 of 88), compared to 47% (17 of 36) for wild hatchlings (results were not statistically tested). After one year in the wild, at least 17 of 32 (53%) head-started hatchling crocodiles were still alive. Authors reported that the released head-started crocodiles adapted well to natural conditions and increased in size. In 2005–2008, crocodile hatchlings were collected from the wild just after hatching (88 individuals) and 32 crocodiles were released back into their natural habitat after being head-started for 14–18 months (31 still held in captivity in 2009). Two ponds (75–450 m^2) were created to provide suitable release habitat.

Growth and survival was monitored by night surveys for one year after release. In 2000–2006, thirty-six wild hatchlings were monitored every three months for one year to compare survival rates.

A study in 2002–2004 on the Narayani and Rapti rivers, Chitwan National Park, Nepal (5) found that approximately half of released captive reared gharials *Gavialis gangeticus* survived at least a year following their release. Nineteen of 36 gharials released survived approximately one year, with two surviving at least two years. Captive reared gharials were released into two different river sections in March 2002 (10 gharials) and March–April 2003 (26 gharials). Individuals were monitored by kayak in November 2002–April 2003 and November 2003–May 2004. Tags and notches on tail scales were used for identification. Gharials were from wild-collected artificially-incubated eggs. Hatchlings were reared until 4–7 years (average body size 1.5 m long) before being released.

A study in 1999–2009 in freshwater and riparian zones in northern Luzon, Philippines (6; same experimental set-up as 4) found that after releasing head-started Philippine crocodiles *Crocodylus mindorensis,* wild crocodile populations in sanctuaries managed by local communities increased in size. Following regular releases of wild-born Philippine crocodiles head-started in captivity, the crocodile population increased to 65 individuals in 2009 from 12 individuals in 2000. The authors reported that survival rates were high (no data are provided) and released crocodiles had no problems adapting to living in the wild. The authors reported that most people in the area knew that crocodiles are legally protected and no crocodiles were killed in the sanctuaries since 2007. After a small population of crocodiles was discovered in 1999, three crocodile sanctuaries were created. Between 2000 (start year not provided) and 2007, wild-born hatchling crocodiles were head-started in captivity for 14 months and then released into the wild. Details of numbers of crocodiles released each year and monitoring were not provided. A communication, education and public awareness campaign about the risks facing crocodiles was carried out in the local rural communities. Crocodile sanctuaries were protected by paid local community members.

A replicated study in 2004–2016 along four rivers in lowland Nepal (7) found that in three of four rivers where head-started gharials *Gavialis gangeticus* had been released, more gharials were counted after eight years and there was evidence of breeding in the wild in all rivers. Results were not statistically tested. In three of four rivers where head-started gharials were released, more gharials were counted in 2016 (Narayani river: 84 individuals; Rapti river: 82 individuals; Babai river: 31 individuals; Karnali river: 1 individual) compared to eight years previously (in 2008, Narayani river: 34 individuals; Rapti river: 23 individuals; Babai river: 10 individuals; Karnali river: 6 individuals). Over the same time period, subadults were observed in all four rivers and hatchlings in three of four rivers (see original paper for details). In 1981–2016, eggs were collected from wild nests and gharials were hatched and head-started in captivity and released aged 4–7 years into the Narayani (~397 individuals), Rapti (~477 individuals), Karnali (~41 individuals) and Babai (~111 individuals) rivers. Gharials were surveyed by boat in November–March over several years between 2004–2016 in Narayani-Rapti rivers or 2008–2016 in Karnali-Babai rivers. Observed gharials were grouped into age classes.

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(2) Muñoz M.D.C. & Thorbjarnarson J. (2000) Movement of captive-released Orinoco crocodiles (*Crocodylus intermedius*) in the Capanaparo River, Venezuela. *Journal of Herpetology*, 34, 397–403.
(3) Larriera A., Siroski P., Pina C.I. & Imhof A. (2006) Sexual maturity of farm-released *Caiman latirosis* (Crocodylia: Alligatoridae) in the wild. *Herpetological Review*, 37, 26–28.
(4) van de Ven W.A.C., Guerrero J.P., Rodriguez D.G., Telan S.P., Balbas M.G., Tarun B.A., van Weerd M., van der Ploeg J., Wijtten Z., Lindeyer F.E. & de Iongh H.H. (2009) Effectiveness of head-starting to bolster Philippine crocodile *Crocodylus mindorensis* populations in San Mariano municipality, Luzon, Philippines. *Conservation Evidence*, 6, 111–116.
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(7) Acharya K.P., Khadka B.K., Jnawali S.R., Malla S., Bhattarai S., Wikramanayake E. & Kohl M. (2017) Conservation and population recovery of gharials (*Gavialis gangeticus*) in Nepal. *Herpetologica*, 73, 129–135.

Tuatara

- **Two studies** evaluated the effects of head-starting wild-caught tuatara for release. Both studies were in New Zealand[1,2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Survival (2 studies):** One study in New Zealand[2] reported that 67–70% of head-started tuatara survived over monitoring periods of 9–11 months. One study in New Zealand[1] found that 56% of head-started tuatara were recaptured over six years following release.
- **Condition (1 studies):** One study in New Zealand[1] reported that head-started tuatara increased in weight by around 100 g during the five years following release[1].

BEHAVIOUR (0 STUDIES)

A study in 1995–2000 on an island in New Zealand (1) found that most head-started tuatara *Sphenodon punctatus* survived at least five years following release. Twenty-eight of 50 head-started juveniles (56%) were recaptured over six years following release, as well as 11 of 18 translocated adults (61%). Juvenile weights increased by approximately 100 g (up to 106% increase) in the five years after release. No successful breeding was observed during the six-year period, though tuatara take 10–15 years to reach maturity. In November 1995, fifty head-started juveniles were released on Titi island (a rodent-free island), along with 18 adults translocated from North Brother Island. Juveniles were selected from those hatched and reared from eggs harvested from the wild population on North Brother Island in 1989–1991. Tuatara were released into artificial burrows at night (2100–2230 h). Six post-release monitoring trips were conducted between November 2995 and November 2000, when a team of 3–4 people spent up to seven nights on the island searching for tuatara.

A study in 2012–2013 in regenerating temperate forest in South Island, New Zealand (2) found that most head-started tuatara *Sphenodon punctatus* survived at least 9 months after being released into a predator-free fenced enclosure with artificial burrows. Results were not statistically tested. After 3–5 months, 100% of tuatara captive-reared locally (13 of 13 individuals) and 96% of tuatara captive-reared to the north of the release site (27 of 28 individuals) had survived. After 9–11 months, 70% of tuatara reared north of the release site (9 of 13 individuals) and 67% of locally-reared and released tuatara had survived (19 of 28 individuals). Juvenile tuatara originating from the same wild population were released into a fenced predator-free reserve in November–December 2012: captive-reared locally to the release site (13 individuals), and captive-reared 480 km north of the release site in a warmer climate (28 individuals). Captive-reared tuatara were hatched from artificially incubated eggs and head started until 4–6 years old. Artificial burrows were buried in the release area. Tuatara were monitored by radio-tracking for 5 months (6 locally-reared, 10 north-reared individuals) and recapture surveys (all tuatara were PIT tagged) for up to 27 months after release.

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(2) Jarvie S., Senior A.M., Adolph S.C., Seddon P.J. & Cree A. (2015) Captive rearing affects growth but not survival in translocated juvenile tuatara. *Journal of Zoology*, 297, 184–193.

14.20. Release reptiles born/hatched in captivity from wild-collected eggs/wild-caught females without rearing

- **Five studies** evaluated the effect on reptile populations of releasing reptiles born/hatched in captivity from wild-collected eggs/wild-caught females without rearing. Four studies were in the USA[2-5] and one was in Australia[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (5 STUDIES)

- **Reproductive success (1 study):** One replicated, controlled study in the USA[2] found that for plains gartersnakes released as newborns, two of over 350 released snakes were found to be gravid two years after release.
- **Survival (5 studies):** One before-and-after study in the USA[5] found that survival of captive-born desert tortoises released as hatchlings was similar over six months compared to hatchlings that were head-started in indoor or outdoor enclosures. One replicated, controlled study in the USA[3] found that alligator hatchlings released into their mother's home range had higher survival than those released outside her home range. Three replicated studies (including one controlled study) in Australia[1] and the USA[2,4] found that 11% of Murray short-necked turtles[1] and 7% of plains gartersnakes[2] survived for 1–3 years after release, and first year survival of gopher tortoise hatchlings released into a predator proof enclosure was around 30%.
- **Condition (2 studies):** One before-and-after study in the USA[5] found that captive-born desert tortoises released as hatchlings grew more slowly over six months than hatchlings head-started in an indoor enclosure. One replicated, controlled study in the USA[3] found

mixed effects on growth of alligator hatchlings released inside or outside of their mother's home range compared to wild hatchlings.

BEHAVIOUR (0 STUDIES)

Background

In some circumstances, it may be desirable to release hatchlings born in captivity from wild-collected eggs or wild-caught females soon after hatching/birth, rather than rearing them in captivity (see *Head-start wild-caught reptiles*).

This action includes studies that discuss the outcomes for hatchlings after they have been released. For studies that discuss the effects of relocating eggs or nests for artificial incubation, or relocating nests to on beach hatcheries, see *Relocate nests/eggs for artificial incubation*; *Relocate nests/eggs to a nearby natural setting (not including hatcheries)* and *Relocate nests/eggs to a hatchery*.

A replicated study in 1996–2000 in two lagoons in south-eastern Australia (1) found that releasing captive-born Murray short-necked turtle *Emydura macquarii* hatchlings from wild-caught females resulted in some surviving for 1–3 years after release. The number of hatchlings that were recaptured over a three-year period was similar at both lagoons (38 of 328, 12% and 30 of 281, 11%) (number of released hatchlings taken from methods). In 1996–1997, gravid female turtles were captured and induced to lay their eggs (number of turtles and method not given). Eggs were artificially incubated, and hatchlings were released in to one of two lagoons (281 and 328 hatchlings each). Fox control was undertaken at one lagoon in May 1997 to January 1999 using poison baits and shooting. Recapture of turtles was carried out in 1998–2000.

A replicated, controlled study in 1995–2001 on an urban river bank with a mix of mown lawns and riparian vegetation in Illinois, USA (2) found that captive-born plains gartersnakes *Thamnophis radix* from wild-caught, gravid females that were released as newborns had low survival in the wild, which was similar to wild-caught newborns. Twenty-seven of 362 (7%) captive-born snakes released as newborns and 2 of 15 (13%) wild-caught newborns were recaptured one or more years after release or initial capture. Seven snakes released as s reached maturity during the study, and two of these were gravid (aged 21–22 months old). In 1995–2001, gravid females were captured (number not given) and maintained in captivity until giving birth. Snakes born in 1998 (137 snakes), 2000 (188 snakes) and 2001 (71 snakes) were released within 2–27 days of birth. Recapture effort varied between months and years, but most snakes were recaptured by hand in April–June 1998–2001.

A replicated, controlled study in 1998–1999 in three lakes in Florida, USA (3) found that captive-born American alligator *Alligator mississippiensis* hatchlings from wild-collected eggs that were released in their mother's home ranges had higher survival than those released outside of their mother's home ranges and similar survival but less growth than wild hatchlings. The chance of recapture after nine months was similar for hatchlings released in their mother's home range (22% recaptured) and wild alligators (23%), whereas hatchlings released outside their mother's home range had a lower chance of recapture (15%). Hatchlings released in their mother's home range were shorter (42 cm) than wild hatchlings (45 cm) nine months after hatching, but hatchlings released outside their mother's home range were similar in length to both other groups (42 cm). Clutches of alligator eggs were collected in summer 1998 from three lakes and artificially incubated at 32°C and hatched. Hatchling alligators were held in captivity for 2–4 weeks before being tagged and released either near to the original nest site (34 clutches) or more than 1 km outside the mother's home range (14 clutches). Wild hatchlings (22 clutches) were collected by hand from boats in September to November 1998, tagged, measured and released. Clutches ranged from 8–41 hatchlings. Alligators were recaptured from May to July 1999 (347 hatchlings from 67 clutches).

A replicated study in 2007–2010 in open mixed pine forest in Georgia, USA (4) found that approximately a quarter of captive-born gopher tortoise *Gopherus polyphemus* hatchlings from wild-collected eggs initially released into predator-proof cages, and then into the wild on an island, survived the first year. Results were not statistically tested. In three consecutive years, survival rates of captive-born hatchling gopher tortoises released into predator-proof cages were 20–29% (178 tortoises in total) in the first year following release. In 2007–2009, gopher tortoise eggs were collected from the wild (from nests or gravid females), or private collections and incubated at 28–30°C. After emergence, 178 gopher tortoise hatchlings were released shortly after hatching into temporary predator-proof release cages (190 cm long x 122 cm wide x 33 cm high, 10–15 individuals/cage) near to abandoned burrows on a 5,670 ha island. Hatchlings remained in cages for two–four weeks before being released. All hatchlings were monitored by live trapping for two weeks in September–October 2007–2010 as well as opportunistically during other trapping exercises in the same years.

A before-and-after study in 2015–2016 in desert scrubland in California, USA (5) found that almost all released captive-born desert tortoise *Gopherus agassizii* hatchlings from wild-collected eggs survived at least six months in the wild, and that hatchlings that were head-started outdoors or indoors had similarly high survival during their head-starting. Survival rates of released hatchlings was similar to that of tortoises during head-starting over six months (released: 15 of 20, 75% survived; during outdoor head-starting: 20 of 20, 100%; during indoor head-starting: 29 of 30, 97%). After six months, released tortoises in the wild were a similar size compared to tortoises during outdoor headstarting (released: 49 mm long; during outdoor head-starting: 51 mm long), but were smaller than tortoises during indoor head-starting (78 mm long). The relative weights and body conditions of tortoises were similar after six months, regardless of rearing approach (see original paper for details). Eggs from 25 wild adult female tortoises were collected, incubated in artificial burrows outside and hatched in August–September 2015. In September 2015, seventy hatchlings (21–46 days old) were either released directly into the wild (20 hatchlings) or moved to either an indoor enclosure (30 hatchlings) or outdoor enclosure (20 hatchlings). Direct-release hatchlings were released in a 0.7 km^2 unfenced area and monitored using radio telemetry twice weekly until November 2015, once a week in winter and twice weekly from March 2016. The indoor enclosure was climate controlled and hatchlings were fed five times/week and watered weekly (see paper for details). The outdoor enclosure (30 x 30 m) was semi-natural, predator-proof and hatchlings were provided supplemental food and water weekly until the end of the active season (November 2015). Hatchling morphometrics were assessed prior to their release or before being moved to their head-starting enclosure in September 2015 and again at least once in March–April 2016.

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(2) King R.B. & Stanford K.M. (2006) Headstarting as a management tool: a case study of the plains gartersnake. *Herpetologica*, 62, 282–292.

(3) Temsiripong Y., Woodward A.R., Ross J.P., Kubilis P.S. & Percival H.F. (2006) Survival and growth of American alligator (*Alligator mississippiensis*) hatchlings after artificial incubation and repatriation. *Journal of Herpetology*, 415–423.

(4) Tuberville T.D., Norton T.M., Buhlmann K.A. & Greco V. (2015) Head-starting as a management component for gopher tortoises (*Gopherus polyphemus*). *Herpetological Conservation and Biology*, 10, 455–471.

(5) Daly J.A., Buhlmann K.A., Todd B.D., Moore C.T., Peaden J.M. & Tuberville T.D. (2018) Comparing growth and body condition of indoor-reared, outdoor-reared, and direct-released juvenile Mojave desert tortoises. *Herpetological Conservation and Biology*, 13, 622–633.

Relocation of nests and eggs

14.21. Relocate nests/eggs to a nearby natural setting (not including hatcheries)

Background

Reptile nests/eggs may be relocated away from specific threats (e.g. egg collecting, flooding, erosion, predation, or being crushed on roads) and reburied in an alternative suitable natural setting where the threat is lower or non-existent. Consideration must be given to the potential impacts of different environmental conditions in the destination location (for example temperature and humidity) on the sex, size, shape, colour, behaviour, movement ability and post-hatching growth of reptile hatchlings (Warner & Andrews 2002, Booth *et al.* 2006).

Due to the number of studies found, this action has been split by species group, though no studies were found for amphisbaenians.

This action does not include studies on the effect of relocating nests/eggs to on-beach hatcheries, which are designated locations on a beach that are often fenced and patrolled, and where larger numbers of nests/eggs tend to be reburied at relatively high densities. Studies on the effect of moving eggs to on-beach hatcheries are discussed in *Relocate nests/eggs to a hatchery*.

For studies on the effect of relocating eggs into artificial settings, including in polystyrene boxes and other containers, see *Relocate nests/eggs for artificial incubation*.

Depending on the threat to nests, practitioners may consider other actions such as *Threat: Invasive alien and other problematic species – Protect nests and nesting sites from predation* and *Threat: Biological resource use – Patrol or monitor nesting beaches*.

See also: *Alter incubation temperatures to achieve optimal/desired sex ratio*.

Booth D.T. (2006) Influence of incubation temperature on hatchling phenotype in reptiles. P*hysiological and Biochemical Zoology*, 79, 274–281.

Warner D.A. & Andrews R.M. (2002) Laboratory and field experiments identify sources of variation in phenotypes and survival of hatchling lizards. *Biological Journal of the Linnean Society*, 76, 105–124.

Sea turtles

- **Thirteen studies** evaluated the effects of relocating nests/eggs to a nearby natural setting on sea turtle populations. Five studies were in the USA[3,7,10,11,13], two were in Suriname[1,2] and the US Virgin Islands[4,12] and one was in each of Costa Rica[5], Ascension Island[6], Brazil[8] and Cape Verde[9].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (12 STUDIES)

- **Reproductive success (12 studies):** Four of 12 controlled studies (including three replicated, randomized studies) in the USA[3,7,10,11,13], Suriname[2], US Virgin Islands[4,12], Costa Rica[5], Ascension Island[6], Brazil[8] and Cape Verde[9] found that relocated sea turtle nests had lower hatching success than natural nests in six of seven years[8], in 26 of 29 years[12], or lower hatching success than nests laid above the tidal zone[6], or that nests relocated >10 days after being laid had lower hatching and emergence success than natural nests or nests relocated within 12 hours[11]. One of those studies[11] also found that relocating nests within 12 hours had mixed effects on hatching and emergence success compared to natural nests. One study[6] also found that two different egg collecting methods resulted in either more dead early stage or late-stage embryos. Four of the studies[5,6,9,10] found that relocated sea turtle nests had similar hatching and emergence success[5,10] or hatching success[7,9] compared to natural nests and specifically compared to those laid in safer parts of the beach[5] or above the high tide line[7]. One of those studies[9] also found that relocated nests experienced similar levels of predation by ghost crabs as natural nests. One of the studies[10] also found that fewer relocated nests failed completely due to tidal flooding compared to natural nests. One of the studies[3] found that relocated loggerhead turtle nests had higher hatching success than natural nests. One of the studies[2] found that relocated leatherback turtle nests had higher hatching success compared to natural nests that were washed over by sea swells, but similar hatching success compared to natural nests that were not washed over by sea swells. The other two studies[4,13] found that relocating sea turtle nests had mixed effects on hatching[4] or hatching and emergence success[13] compared to natural nests. One of those studies[4] also found that in years when leatherback turtle nests were relocated, fewer were lost to erosion than when no relocations took place.
- **Condition (1 study):** One replicated, randomized, controlled study in the USA[7] found that hatchlings from relocated loggerhead turtle nests were a similar size to hatchlings from natural nests.

BEHAVIOUR (0 STUDIES)

OTHER (1 STUDY)

- **Offspring sex ratio (1 study):** One replicated, randomized, controlled study in Suriname[1] found that relocated leatherback turtle nests produced all female hatchlings, whereas 30–100% of hatchlings from naturally incubated nests were female.

A replicated, controlled study in 1980–1982 on a sandy beach in Suriname (1; same experimental set-up as 2) found that moving leatherback turtle *Dermochelys coriacea* nests to above the tideline produced all female hatchlings, whereas natural nests produced mixed sex ratios and artificially incubating in Styrofoam boxes produced all male hatchlings. Leatherback turtle hatchlings reburied in the sand on the same beach produced 100% female hatchlings, compared to 30–100% of female hatchlings in natural nests and 100% male hatchlings in Styrofoam-box-incubated nests. In 1982, leatherback turtle eggs from two clutches laid below the tide line were reburied elsewhere on the beach. Ten hatchlings were randomly selected after emergence, euthanised and sexed. Sex ratios were compared to 10 hatchlings/clutch of two naturally-incubated nests laid in 1980, six naturally-incubated nests laid in 1982 and five clutches in 1980 and 10 clutches in 1982 incubated in Styrofoam boxes (45–60 eggs/box).

A replicated, controlled study in 1982 on a sandy beach in Suriname (2; same experimental set-up as 1) found that leatherback turtle *Dermochelys coriacea* nests reburied above the tide line had similar hatching success but lower predation rates than natural nests not washed over by sea swells. Results were not statistically tested. Average hatching success of leatherback turtle eggs reburied above the tide line was 69% compared to 33% in natural nests washed over by sea swells and 62% in natural nests not washed over by sea swells. Embryonic mortality in reburied nests was 23% compared to 35% and 13% in natural nests washed over and not washed over by swells respectively. Predation rates were 6% in reburied nests compared to 27% and 17% in natural nests washed over and not washed over respectively. Nesting turtles were surveyed <once/week in March–August 1982 on a 12 km long beach. Nests laid below the spring high tide line were relocated the next day to further up the beach above the spring high tide line (13 leatherback clutches, 50 eggs/clutch reburied together in 60 cm deep cavities 1 m apart). Relocated and natural nests (10–12 nests occasionally washed over by sea swells, 13–16 not washed over by sea swells) were excavated after emergence to evaluate hatching success.

A replicated, controlled study in 1984 on a sandy beach in Florida, USA (3) found that relocating loggerhead turtle *Caretta caretta* nests to other locations on the beach or for artificial incubation resulted in higher hatching success compared to nests left in place (but protected from predation). Hatching success was higher for relocated nests (1,054 of 1,151 eggs, 92% hatched from 10 nests) compared to nests left in place (2,400 of 2,796 eggs, 87% hatched from 24 nests). Hatching success did not differ for relocated nests that were reburied (543 of 588, 92% hatched from 5 nests) or artificially incubated (511 of 563 eggs, 91% hatched from 5 nests). Six nests left in place were lost to the tide or vandalism. Ten nests at risk from predation or tides were reburied in another part of the beach (five clutches) or were incubated in polystyrene boxes with sand (38 x 38 x 19 cm). A further 31 nests were screened to prevent predation and left in place. Hatching success was assessed following emergence of hatchlings.

A controlled, before-and-after study in 1981–1994 on a sandy beach in St Croix, US Virgin Islands (4) found that relocating leatherback turtle *Dermochelys coriacea* nests away from erosion-prone areas lead to fewer nests being lost to erosion compared to when no nests were relocated, and variable hatching success in relocated compared undisturbed nests. Results were not statistically tested. In years when nests were relocated, 1–30% were lost to erosion (of 82–355 nests), whereas 48 of 119 (40%) were lost in the year in which no relocations took place. Hatching success was lower in relocated nests (51–69%) compared to undisturbed nests (57–76%) in 10 of 13 years. In 1982–1994, all nests in erosion-prone areas were relocated to stable parts of the beach immediately after laying. In 1981–1994 the beach was patrolled hourly between 20:00–05:00 h every night from 1st April until no new nests had not been discovered for 10 days. Nests were excavated several days following emergence to record hatching success.

A replicated, controlled, before-and-after study in 1999–2004 on a sandy beach in Guanacaste Province, Costa Rica (5) found that relocating leatherback turtle *Dermochelys coriacea* nests to safe locations on the beach or a hatchery resulted in similar hatching and emergence success compared to nests left in situ. Results were not statistically tested, and no distinction was made between nests relocated to the beach or hatchery. Hatching and emergence success were similar for relocated nests (hatching: 19–52%; emergence: 14–32%) and nests left in situ (hatching: 30–69%; emergence: 9–57%). In October–March 1999–2004, beaches were searched nightly for nesting females. In 2001–2004, nests considered to be at high risk (such as being within tidal zone, in areas of high pedestrian traffic, in vegetation or close to estuary) were relocated to safe places on the beach or a hatchery (86 nests), and others were left in situ (220 nests). Two days after emergence of the first hatchling, or 60 days after laying, nests were excavated to determine hatching and emergence success.

A replicated, randomized, controlled study in 2006 on a sandy beach on Ascension Island (6) found that relocating green turtle *Chelonia mydas* eggs from nests in the tidal zone resulted in lower hatching success compared to in situ nests laid above the tidal zone. Hatching success was lower for relocated nests (66% and 67%) compared to natural nests laid further up the beach (86%). One relocation method (collecting eggs during laying process) resulted in more early-stage dead embryos compared to in situ nests (22 vs 9/nest) and the other method (nest excavation) in more late-stage dead embryos (17 vs 7/nest). In March–April, a 1 km stretch of beach was searched for nesting females. Nests in the tidal area (doomed nests) were relocated close to one of 23 natural nests laid further up the beach (23 locations further up the beach, with 2 relocated nests and 1 natural nest/location). Eggs were relocated by excavating the nest following completion of nesting; or by removing eggs from the chamber during the laying process. After hatchling emergence nests were excavated to assess hatching success.

A replicated, randomized, controlled study in 2005–2006 on a beach in Georgia, USA (7) found that relocating loggerhead turtle *Caretta caretta* nests to areas above the high-tide resulted in higher hatching success compared to nests laid above and below the high-tide that were left in place. Hatching success was higher for relocated nests (81%) than natural nests overall (above and below high tide line; 61%) but was statistically similar to natural nests laid above the high tide (72%). Overall, hatching success was higher above the high tide (79%) than below

(54%). There was no significant difference between relocated and natural nests for incubation duration (relocated: 55 days; natural: 54 days) or hatchling size (relocated: 45 mm; natural: 45 mm). In May–August 2005–2006, turtle nests were randomly selected to be relocated (34 nests) or left in place (35 nests). Relocated nests were reburied above the high tide line in nests dug to match the dimensions of the original. All nests were covered with a metal screen to prevent predation. Following hatchling emergence, hatching success was determined and 20 hatchlings were selected for measuring.

A replicated, controlled study in 2004–2011, along 100 km of sandy beach in Rio de Janeiro State, Brazil (8) found that relocating loggerhead turtles *Caretta caretta* nests to nearby locations on the beach resulted in lower hatching success compared to nests left in situ. Hatching success was lower for relocated nests than for nests left in situ in six of seven seasons (relocated: 57–69%; in situ: 73–81%). In addition, hatching success was also lower for nests relocated to an on-beach hatchery in six of seven seasons (61–66%) compared to in situ nests. In the nesting seasons of 2004–2011 beaches were patrolled daily, and nests were transferred to a safe location on the beach (24–172 nests/season); moved to an on-beach hatchery (231–1,015 nests/season); or left in situ (8–316 nests/season). Those nests not taken to the hatchery were covered with a wire mesh screen. After hatchling emergence, nests were excavated to assess hatching success.

A controlled study in 2008 on a sandy beach in Boa Viste, Cape Verde (9) found that loggerhead turtle *Caretta caretta* nests relocated away from the shoreline experienced a similar amount of predation by ghost crabs *Ocypode cursor* and had similar hatching success compared to natural nests left in situ. Ghost crab predation rates were similar in nests that were relocated away from the shoreline (41%) and nests that were left in place (55%). Hatching success was also similar in nests that were relocated away from the shoreline (42% success) and nests that were left in place (33% success). Turtle nests were excavated, eggs counted and reburied in another part of the beach (20 nests) or left in the same place without any protection (20 nests). Nests were monitored daily until emergence and hatchling tracks were counted. All nests were excavated after last emergence and remaining eggs counted for analysis.

A replicated, randomized, controlled study in 2002–2007 on two sandy beaches in Georgia, USA (10) found that relocating loggerhead turtle *Caretta caretta* nests resulted in similar hatching and emergence success and fewer nests being flooded compared to nests left in situ. When accounting for nest elevation, hatching and emergence success were similar for relocated nests (hatching: 70–73%; emergence: 67%) and nests left in situ (hatching: 76–80%; emergence: 68–78%). Fewer relocated nests failed completely than in situ nests (relocated: 13 of 168, 8%; in situ: 44 of 212, 21%; not statistically tested) and more relocated nests avoided tidal flooding (relocated: 94–98%; in situ: 71–81%; not statistically tested). Two stretches of beach (3 and 7 km) were searched daily during May–October 2002–2007. Nests were either relocated to the top of a nearby dune (85 with a plastic screen; 83 no screen) or were left in situ (75 screened; 137 with no screen). Data from 2004 were excluded due to tropical storms. Nests were excavated five days after hatchling emergence began and the numbers of hatched and unhatched eggs and live or dead hatchlings were counted.

A controlled study in 2007 on a sandy beach in Florida, USA (11) found that relocating loggerhead turtle *Caretta caretta* nests more than 10 days after being laid reduced hatching and emergence success. Loggerhead turtle nests relocated >10 days after being laid had lower hatching (52%) and emergence success (47%) compared to nests relocated within 12 hours to native sand (hatching success:79%; emergence success: 68%) or restored beach (hatching success: 90%; emergence success: 87%), or compared to nests left in situ (hatching success: 85%; emergence success: 84%). Nests relocated within 12 hours to restored beach and native sand had statistically similar hatching success to nests left in situ, but emergence success of nests relocated to native sand was statistically lower than nests left in situ or relocated to restored beach. In May-June 2007, as part of post-storm beach restoration, 12 loggerhead turtle nests (1,429 eggs) were moved 10–38 days after being laid to a section of the beach with native sand. All new nests that were laid in the restoration zone were moved within 12 hours of deposition to native sand beach (63 nests; 7,563 eggs) or restored beach (43 nests; 5,155 eggs). Nests laid on the beach after restoration was complete were left in situ (86 nests; 9,921 eggs). All nests were monitored for hatching and emergence success.

A replicated, controlled study in 1982–2010 on a sandy beach in St Croix, US Virgin Islands (12; continuation of 4) found that relocated leatherback turtle *Dermochelys coriacea* nests had lower hatching success than nests left in situ in 26 of 29 years. In 1982–1994, hatching success was lower in relocated nests (51–69%) compared to undisturbed nests (57–76%) in 10 of 13 years (result not tested statistically). In 1995–2010, hatching success was lower in relocated nests (37–66%) than nests left in situ (43–69%) every year. In 1982–2010, all nests in erosion-prone areas were relocated to stable parts of the beach immediately after laying. The beach was patrolled hourly between 20:00–05:00 h every night from 1 April until no new nests had been discovered for 10 days. Nests were excavated several days following emergence to record hatching success.

A replicated, controlled study in 2016 on one sandy beach in Alabama, USA (13) found that relocating loggerhead turtle *Caretta caretta* nests higher up the beach resulted in similar hatching success, but lower emergence success compared to undisturbed nests. Hatching success was similar for relocated (66%) and undisturbed nests (66%), but relocated nests had lower emergence success (relocated: 76%; undisturbed: 84%). Seven measures of flooding and wave wash-over were similar at the locations of relocated nests, original nest locations and undisturbed nests. In May–August 2016, twenty nests discovered 0–22 m from the high-tide line were relocated higher up the beach. Seventy-four nests (3–50 m above high-tide line) were left undisturbed. Nest locations were monitored for up to 75 days and the fate of eggs was checked three days after hatchling emergence, or 75 days after laying occurred.

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(11) Ahles N. & Milton S.L. (2016) Mid-incubation relocation and embryonic survival in loggerhead sea turtle eggs. *The Journal of Wildlife Management*, 80, 430–437.
(12) Garner J.A., MacKenzie D.S. & Gatlin D. (2017) Reproductive biology of Atlantic leatherback sea turtles at Sandy Point, St. Croix: the first 30 years. *Chelonian Conservation and Biology*, 16, 29–43.
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Tortoises, terrapins, side-necked & softshell turtles

- **Four studies** evaluated the effects of relocating nests/eggs to a nearby natural setting on tortoise, terrapin, side-necked & softshell turtle populations. One study was in each of Venezuela[1], Columbia[2], Canada[3] and the USA[4].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (4 STUDIES)

- **Reproductive success (4 studies):** Two of four replicated, controlled studies in Venezuela[1], Columbia[2], Canada[3] and the USA[4] found that relocated Arrau turtle[1] and Magdalena river turtle[2] nests had similar hatching success compared to natural nests[1,2]. One of the studies[3] found that painted turtle and snapping turtle nests relocated to artificial nest mounds had higher hatching success than natural nests. The other study[4] found that relocating diamondback terrapin nests to artificial nest mounds had mixed effects on hatching success compared to natural nests.

- **Survival (1 study):** One replicated, controlled study in Venezuela[1] found that Arrau turtle hatchlings from relocated nests had lower survival during their first year compared to hatchlings from natural nests.

- **Condition (2 studies):** One replicated, controlled study in Venezuela[1] found that Arrau turtle hatchlings from relocated nests had more physical abnormalities compared to hatchlings from natural nests. One replicated, controlled study in Columbia[2] found that a similar number of eggs were infested by invertebrates and fungi in relocated and natural nests.

BEHAVIOUR (0 STUDIES)

A replicated, controlled study in 2003 on one river in Venezuela (1) found that relocating Arrau turtle *Podocnemis expansa* nests led to no difference in hatching success, but higher mortality during a year in captivity compared to turtles from naturally incubated nests. There was no significant difference in hatching success between relocated and natural nests (54–98%), but mortality during the first year was higher for turtles from relocated nests (relocated: 13 of 108, 12%; natural: 1 of 112, <1%). At two location on the shell, relocated turtles had more physical abnormalities than naturally incubated turtles (relocated: 74% and 77%; naturally incubated: 19% and 33%), whereas at a third location the number of physical abnormalities was similar (relocated: 4%; naturally incubated: 5%). In February 2003, six nests were excavated and reburied 1.5 km further up the riverbank. In April 2003, hatchlings from the relocated nests, as well as hatchlings from four naturally incubated nests were collected and moved to captivity. A total of 230 turtles (up to 28 turtles/nest) were included in the study and kept in captivity for up to a year.

A replicated, controlled study in 2005–2006 in one wetland and two riverbank sites in northern Columbia (2) found that hatching success of Magdalena river turtle *Podocnemis lewyana* eggs was similar in relocated, artificial and natural nests. Hatching success was statistically similar in relocated nests (58%), artificial nests (21%) and natural nests (41%). The number of eggs infested by invertebrates and fungi was statistically similar for relocated and artificial nests (34%) and natural nests (35%). In 2005–2006, twenty-four nests were relocated higher up the beach away from rising river levels, and seven artificial nests were dug for eggs recovered from turtles that had been harvested by people. A further 22 nests were left in place. All nests were covered with wire mesh cylinders (1 x 1 cm) that were 40 cm wide and 50 cm high, with a 3 x 3 cm plastic mesh on top. In February–May 2005–2006, beaches were searched daily, with the aid of dogs *Canis lupus familiaris*, to locate turtle nests. All nests were inspected daily and excavated after hatching, or after 74 days of incubation.

A replicated, controlled study in 2009–2010 in a mosaic of wetlands, rivers and lakes in Ontario, Canada (3) found that relocating painted turtle *Chrysemys picta* and snapping turtle *Chelydra serpentina* eggs to artificial mounds resulted in higher hatching success than for eggs left in natural nests. Eggs transplanted to artificial nests had higher hatching success than those left in natural nests for nine painted turtle nests (artificial: 98%; natural 71%) and 12 snapping turtle nests (artificial: 88%; natural 56%). Four artificial nesting mounds (60% gravel and 40% sand) 6m diameter and 0.5 high were installed in April 2009 on top of a layer of geotextile cloth. Each mound was within 100 m of water, 50 m of a known nesting site and sited to prevent nesting turtles from having to cross a road. Nests were excavated and split evenly between the closest artificial mound and the original nest. Hatching events were monitored from August, and nests were excavated in October to assess hatching success.

A replicated, controlled study in 2006–2007 on an island on salt marsh grasses in New Jersey, USA (4) found that relocating diamondback terrapin *Malaciemys terrapin* nests to artificial nest mounds resulted in lower hatching success compared to natural nests in three of 12 comparisons. Hatching success for relocated nests ranged from 0–85%, and for natural nests it was 54% and 70%. Hatching success was lower in dredge soil (0%) and shaded sand (0%) compared to natural nests (54%) in the first year, but all other comparisons found no significant differences. Three experimental plots (2.25 m^2) were filled with 45 cm of sand/soil: sand from a beach; loamy sand from a natural nesting area or dredge soil from a nearby channel which had been dried for two months. One half of each plot was shaded by shade cloth 15 cm above the soil with the other half in full sun and each nest had a predator excluder made of wire mesh. Natural nests were in full sun with nearby vegetation cover. Clutches were relocated to treatment plots from areas with high human activity (2006: 5 nests/treatment, 5 natural nests; 2007: 6 nests/treatment, 8 natural nests). Nests were excavated after 60 days to assess hatching success.

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Snakes & lizards

- We found no studies that evaluated the effects of relocating nests/eggs to a nearby natural setting on snake and lizard populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Crocodilians

- We found no studies that evaluated the effects of relocating nests/eggs to a nearby natural setting on crocodile populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Tuatara

- We found no studies that evaluated the effects of relocating nests/eggs to a nearby natural setting on tuatara populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

14.22. Relocate nests/eggs to a hatchery

Background

Reptile nests may be relocated away from specific threats (e.g. egg collecting, flooding, erosion, predation, or being crushed on roads) and reburied in an organised 'hatchery'. Hatcheries consist of a defined location on or near the nesting beach, well above the high tide line, that is often fenced and patrolled. Nests/eggs collected from the beach are then reburied within the hatchery, where they can be closely monitored.

Burying a potentially large number of nests/eggs within a relatively small area may present a number of risks, and the consequences of disturbances such as flooding or poaching could be particularly severe. Other environmental variables at the hatchery location (e.g. temperature and humidity) may also impact on the sex, size, shape, colour, behaviour, movement ability and post-hatching growth of the hatchlings (Warner & Andrews 2002, Booth *et al*. 2006), and should be carefully considered when selecting the location.

Due to the number of studies found, this action has been split by species group, though no studies were found for amphisbaenians.

For studies that discuss moving nests/eggs to other locations on the beach, but not to a hatchery, see *Relocate nests/eggs to a nearby natural setting (not including hatcheries)*, and for those that discuss the effects of relocating eggs into artificial settings, including into polystyrene boxes and other containers, see *Relocate nests/eggs for artificial incubation*.

See also: *Alter incubation temperatures to achieve optimal/desired sex ratio*.

Booth D.T. (2006) Influence of incubation temperature on hatchling phenotype in reptiles. P*hysiological and Biochemical Zoology*, 79, 274–281.

Warner D.A. & Andrews R.M. (2002) Laboratory and field experiments identify sources of variation in phenotypes and survival of hatchling lizards. *Biological Journal of the Linnean Society*, 76, 105–124.

Sea turtles

- **Twenty-two studies** evaluated the effects on sea turtle populations of relocating nests/eggs to a hatchery. Four studies were in each of Malaysia[2,3,15a,15b], Mexico[5,11,19,20] and Costa Rica[9,12,16,18], three studies were in Brazil[4,10,13], two studies were in Cape Verde[8,17] and one study was in each of the USA[1], Turkey[6], Greece[7], Indonesia[14] and Mauritius[21].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (22 STUDIES)

- **Reproductive success (19 studies):** Four of 10 studies (including seven replicated, controlled studies) in Brazil[4,10,13], Mexico[5], Greece[7], Cape Verde[8,17], Costa Rica[9,16], Indonesia[14] found mixed effects on hatching success in sea turtle nests relocated to hatcheries compared to natural nests[4,7,8,10]. Three studies[5,9,16] found that sea turtle nests relocated to hatcheries had similar hatching or emergence success compared to natural nests, and specifically those laid in safe locations[9] or those that were camoflaged[16]. Two studies[14,17] found that nests relocated to hatcheries had higher hatching success than natural nests, and in one case all the natural nests were predated[14]. The other study[13] found that nests relocated to a hatchery had lower hatching success than natural nests in six of seven seasons. Two of the studies[5,17] also found that fewer nests relocated to hatcheries were lost to erosion[5] or predation[5,17] compared to natural nests. One of the studies[8] also found that hatching success was similar following immediate relocation compared to delayed but careful relocation. Four studies (including one replicated, randomized study) in Malaysia[3], Mexico[11], Costa Rica[18] and Mauritius[21] reported that hatching success of sea turtle eggs and nests relocated to hatcheries ranged from 35–78%. One study[3] also found that hatching success was not affected by the number of eggs in the nest. Three studies (including one randomized replicated study) in the USA[1], Malaysia[2] and Mexico[19] found that sea turtle nests relocated to hatcheries had similar hatching success compared to those relocated for artificial incubation. One study[2] also found that handling eggs during the first five days did not affect hatching success. One replicated, controlled, before-and-after study in Costa Rica[12] found that leatherback turtle nests relocated to a hatchery or to other parts of the beach (results combined) had similar hatching success compared to natural nests. One replicated, controlled study in Turkey[6] found that hatching success was similar if nests were relocated 0–18 h after laying[6].
- **Survival (2 studies):** Two studies in Costa Rica[18] and Mauritius[21] found that 77% of olive ridley turtle hatchlings[18] and 89% of green turtle hatchlings[21] from hatcheries successfully reached the ocean.
- **Condition (4 studies):** Two randomized studies (including one replicated, controlled, before-and-after study) in Mexico[19,20] found that relocating olive ridley turtle nests to a hatchery had mixed effects on size[20] or size, movement and condition[19] of hatchlings compare to hatchlings that were artificially incubated[19] or from natural nests[20]. One study[20] also found that hatchery hatchlings had higher stress hormone levels than hatchlings from natural nests after emergence, and a different stress response to reaching the ocean compared to hatchlings from natural nests. One replicated, randomized study in Malaysia[15a] found that green turtle hatchlings released from hatcheries immediately after emergence moved faster than hatchlings held in the hatchery for 1–6 hours and had better body condition than hatchlings held for 3–6 hours. One replicated study in Malaysia[15b] found that excavating green turtle hatchlings in a hatchery immediately after the main clutch emerged resulted in better movement and body condition compared to hatchings excavated five days later.

BEHAVIOUR (0 STUDIES)

OTHER (1 STUDY)

- **Offspring sex ratio (1 study):** One replicated, randomized study in Malaysia[3] found that all but 1 of 169 leatherback turtle eggs relocated to a hatchery produced female hatchlings.

A replicated study in 1983 on a sandy beach in Georgia, USA (1) found that relocating loggerhead turtle *Caretta caretta* nests to a hatchery resulted in similar hatching success compared to eggs relocated for artificial incubation. Hatching success was similar for eggs relocated to a hatchery (3,608 of 5,100, 71% of eggs hatched) and artificially incubated eggs (135 of 163, 83% of eggs hatched). Nine of 50 relocated clutches (18%) were partially destroyed by ghost crab *Ocypode quadrata* predation, cold weather or drifting sand. In 1983, all loggerhead turtle nests on one beach (53 nests) were relocated due to risk of total failure (due to predators, storm tides or poachers). Fifty clutches were reburied in hand-dug nests in a fenced area on a nearby dune and three clutches were placed in glass-fronted polystyrene incubators (38 x 38 x 19 cm). Hatching success was assessed after hatchlings emerged.

A replicated, randomized, study 1986 on a sandy beach in Rantau Abang, Malaysia (2; same experimental set-up as 3) found that relocating leatherback turtle *Dermochelys coriacea* eggs to an on-beach hatchery resulted in similar hatching success compared to eggs that were incubated artificially in Styrofoam boxes. Hatching success was similar for eggs from the hatchery (13–92% of 23–25 eggs) and eggs from Styrofoam boxes (52–100% of 23–25 eggs). In addition, careful handling of eggs during the first five days of incubation did not affect hatching success (handled eggs: 70–100%; non-handled eggs: 52–100%). Eggs were collected from four natural nests (only yolked eggs of normal size) and four groups of eggs (23–25 eggs/group) were incubated in one of three treatments: an on-beach hatchery; in Styrofoam boxes with egg handling during the first five days; or in Styrofoam boxes with no handling (98 eggs/treatment). Eggs in the on-beach hatchery were buried 60 cm deep, and the nests were surrounded with chicken mesh after 50 days. Half of the Styrofoam boxes were kept in a well-ventilated shed, and the others were kept in an enclosed laboratory. Hatching success was measured by counting the number of hatchlings that emerged.

A replicated, randomized study in 1986 on a sandy beach in Rantau Abang, Malaysia (3; same experimental set-up as 2) found that moving leatherback turtle *Dermochelys coriacea* eggs to an on-beach hatchery resulted in almost all female hatchlings, and that hatching success did not vary due to the number of eggs buried together in the hatchery. In total, 168 of 169 (99%) hatchlings were female, and the sex of one hatchling could not be determined. Hatching success was similar across all nest sizes (23–25 eggs: 13–48%; 48–50 eggs: 22–44%; 72–75 eggs: 9–51%; 96–100 eggs: 15–47%). In 1986, eggs were collected from a total of 13 natural nests and buried in groups of 23–25 (7 nests), 48–50 (3 nests), 72–75 (3 nests) or 96–100 eggs/nest (3 nests). From each nest, 2–20 hatchlings were selected for sexing. These hatchlings were euthanised in chloroform and sex was determined by removing and examining the gonads. Hatching success was determined by counting the number of hatchlings to emerge from each nest.

A replicated, controlled study in 1987–1993 on a beach in Bahia, Brazil (4), found that relocating sea turtle nests to an on-beach hatchery resulted in lower hatching success compared to nests left in situ (though some of these nests were protected or moved) for one of two species. Hatching success was lower for loggerhead turtle *Caretta caretta* nests in the hatchery (63%) than for nests left in situ (73%), though there was no significant difference for hawksbill turtles *Eretmochelys imbricata* (hatchery: 52%; left in situ: 61%). Hatching success was higher for loggerhead nests relocated within six hours of laying (69%) than for nests relocated more slowly (6–12 hours: 63%; >12 hours: 63%), but there was no significant difference for hawksbills (within six hours: 57%; 6–12 hours: 53%; >12 hours: 56%). In September–May 1987–1993, three sections of a beach were patrolled daily to record nesting events. Eggs from 1,659 nests on two sections of the beach (19 and 10 km long) were brought to a fenced hatchery located on a third section of the beach (14 km), where they were reburied (15 cm deep) and surrounded by plastic mesh cylinders (35 cm high, 60 cm wide). A further 514 nests were left in situ, but those at risk from predation were covered with a plastic mesh (100 x 100 cm), and those at risk from tidal flooding or human activity were relocated to another natural setting on the beach (number of nests not reported).

A replicated study in 1988–1997 on a sandy beach in Jalisco, Mexico (5) found that relocating olive ridley turtle *Lepidochelys olivacea* nests to an on-beach hatchery resulted in fewer nests being lost to erosion or predation, and similar hatching success compared to nests left in situ. None of 65 relocated nests were lost to erosion or predation, whereas only 36 of 65 (56%) nests left in situ survived. Hatching success was similar in relocated (59%) and in situ nests (66%). In August 1990, July 1991 and October 1994, a 3 km stretch of beach was patrolled for nesting turtles. Half of the nests discovered were relocated to the hatchery and half were left in situ (total of 18, 32 and 80 nests/year). The hatchery (10 x 35 m) was enclosed with a fence made of wire mesh and mosquito mesh (2.5 m high, 0.5 m deep). Hatchlings were counted and released on the beach, and nests were excavated to assess hatching success.

A replicated, controlled study in 2000–2002 on three sandy beaches in southwest Turkey (6) found that relocating sea turtle nests to an on-beach hatchery may have resulted in higher hatching success compared to nests left in situ. Results were not statistically tested. Hatching success tended to be higher for relocated nests (89, 85 and 71% hatching success for 5, 37 and 6 nests respectively) compared to nests left in situ (68, 19 and 64% for 67, 40 and 97 nests respectively). For relocated nests, hatching success was not affected by time after laying that relocation took place (0–6 h: 89%; 6–12 h: 79%; 12–18 h: 70%). Nests considered to be in vulnerable locations were relocated to a hatchery. The hatchery (10 x 15 m) was enclosed by a plastic fence (2 cm mesh) that extended 0.5 m below and 2 m above the sand surface. Hatchlings were released manually from near original nest sites. Nests were monitored from June–September 2000–2002.

A replicated, controlled study in 1987–1995 on a sandy beach on Zakynthos Island, Greece (7) found that relocating loggerhead turtles *Caretta caretta* nests to an on-beach hatchery resulted in variable hatching success compared to both nests left in situ and nests left in situ and covered with metal cages. Average hatching success in the on-beach hatchery varied from 51–75%, compared with 56–68% for in situ nests and 44–72% for in situ nests covered with cages. Hatching success in the hatchery was lower in one of eight years and higher in two of eight years compared to in situ nests. From 1988, nests located within 7 m of the sea and in danger of inundation were moved to a beach hatchery (77 nests) as were nests located near to invasive plants with root systems that may have grown into nests. From 1990, nests located in beach areas with tourists were protected by 50 cm circular metal mesh cages buried 15 cm in the sand (88 nests). A further

313 nests were left in situ. Nests were excavated following hatchling emergence to assess hatching success.

A replicated, controlled study in 2005 on Boavista Island, Republic of Cabo Verde, western Africa (8) found that relocating loggerhead turtle *Caretta caretta* eggs to an on-beach hatchery resulted in lower egg mortality than naturally incubated eggs from one of two beaches, and that delayed, careful relocations resulted in similar mortality compared to immediate egg relocations. Egg mortality was lower for hatchery nests (immediate non-careful relocation: 38%; delayed and/or careful relocation: 48%) compared to natural in situ nests on one other beach (79%), but similar to natural in situ nests on another beach (56%). Egg mortality was similar for immediate (38%) and delayed, careful (48%) relocation, and mortality was similar regardless of the length of the time delay (0–96 h after laying: 41–55% mortality). Eggs relocated to the on-beach came from nests laid in flood-prone or silty areas. Eggs from 50 nests were moved at 0, 12, 24, 84, and 96 post-laying (10 nests/treatment), and care was taken to keep eggs upright. Eggs from a further 134 nests were taken to the hatchery immediately after laying with no care taken to control egg vibration or orientation. Eggs from two other beaches (41 and 34 nests each) were left in the nests to incubate naturally. All nests were excavated five days after the last emergence to assess egg mortality.

A replicated, controlled study in 1990–2004 on one sandy beach on the Caribbean coast of Costa Rica (9) reported that relocating leatherback turtle *Dermochelys coriacea* nests from areas at high-risk of erosion to an on-beach hatchery resulted in similar emergence success compared to nests in low-risk areas. Results were not statistically tested. Emergence success was similar for eggs in the hatchery (43%) compared to eggs from nests in low-risk areas of the beach (41%). In February–July 1990–2004, all nests laid in high-risk areas (within 100 m of a river mouth) were relocated to on-beach hatcheries. Hatcheries were fenced and staffed 24 h/day during the incubation period, and all reburied nests were surrounded by a metal mesh cylinder to exclude predators and a fine cloth mesh (1 mm) to exclude flies. In February–July 1990–2004, nest surveys were conducted every night between 20:00–04:00 h, and all nests were monitored four times/day.

A replicated, controlled study in 1991–2003 on eight sandy beach locations in Sergipe and Bahia, Brazil (10) found that relocating olive ridley turtle *Lepidochelys olivacea* eggs to a hatchery resulted in similar hatching success compared to nests left in situ in seven of 12 nesting seasons and lower hatching success in five of 12 seasons. In seven of 12 nesting seasons, hatching success was similar for nests in the hatchery (76–84%) and nests left in situ (76–85%). In five seasons hatching success was lower in on-beach hatcheries (73–80%) than in situ nests (81–85%; see original paper for details). Turtle nesting activity was monitored on eight stretches of beach (339 km total length) in 12 nesting seasons (September–March) from 1991/1992–2002/2003. Hired fishers surveyed the beaches every morning to locate, count and move nests at risk from a range of threats (including tidal inundation, predators, poaching, beach illumination or habitat alteration) to open beach hatcheries or other areas of beach. Nests not at risk were left in situ. Hatchery and in situ nests were excavated after emergence to determine clutch size and hatching success (hatchery: 160–969 olive ridley nests/year; in situ: 7–286 olive ridley nests/year).

A study in 1982–2005 on four beaches on the pacific coast of Mexico (11) reported that after relocating leatherback turtle *Dermochelys coriacea* nests to an on-beach hatchery, some successfully hatched. Results were not statistically tested. Average hatching success/year was 35–53%. Over the course of the study, at least 639,000 eggs were moved to the hatchery, and at least 270,000 hatchlings were released in to the wild. Patrols of at least one of four beaches took place annually in 1982–2005. In 1997–2005, the survey protocol was standardized across all four beaches, and nightly patrols to search for nests took place in October–May at 20:00–05:00 h. Clutches of eggs were gathered as soon as possible after laying (normally within 1–2 hours) and reburied in a protected, fenced area. Hatchlings were released on emergence at a number of different locations on the beach.

A replicated, controlled, before-and-after study in 1999–2004 on a sandy beach in Guanacaste Province, Costa Rica (12) found that relocating leatherback turtle *Dermochelys coriacea* nests to a hatchery or to other safe locations on the beach resulted in similar hatching and emergence success compared to nests left in situ. Results were not statistically tested, and no distinction made between nests relocated to the hatchery or to other locations on the beach. Hatching and emergence success were similar for relocated nests (hatching: 19–52%; emergence: 14–32%) and nests left in situ (hatching: 30–69%; emergence: 9–57%). In October–March 1999–2004, beaches were searched every night for nesting females. In 2001–2004, nests considered to be at high risk (within tidal zone, in areas of high pedestrian traffic, in vegetation, close to estuary) were relocated to a hatchery or to other safe places on the beach (86 nests), and other nests were left in situ (220 nests). Two days after emergence of the first hatchling, or 60 days after laying, nests were excavated to determine hatching and emergence success.

A replicated, controlled study in 2004–2011 along 100 km of sandy beach in Rio de Janeiro State, Brazil (13) found that relocating loggerhead turtles *Caretta caretta* nests to an on-beach hatchery resulted in lower hatching success compared to nests left in in situ. Hatching success was lower for hatchery nests than for nests left in situ in six of seven seasons (hatchery: 61–74%; in situ: 73–81%), and similar in one season (hatchery: 65%; in situ: 79%). In the nesting seasons of 2004–2011 beaches were patrolled daily, and nests were either moved to an on-beach hatchery (231–1,015 nests/season) or left in situ (8–316 nests/season). After hatchling emergence, nests were excavated to assess hatching success.

A randomized, controlled study in 2009–2010 on a sandy beach in East Java, Indonesia (14) found that relocating olive ridley turtle *Lepidochelys olivacea* nests to an on-beach hatchery resulted in higher hatching success than for nests left in situ. The hatching success of nests moved to an on-beach hatchery was 54–73% of nests laid (2009: 39 of 53 nests; 2010: 30 of 56), whereas all nests left in place (11–19 nests/year) were lost to predation within one week of being laid and no eggs hatched. Olive ridley turtle nests laid in May–July 2009–2010 along an 18 km stretch of sandy beach in a national park were randomly selected to be moved to an on-beach hatchery (2009: 53 nests; 2010: 56 nests) or left in place (2009: 11; 2010: 19) within 200 m of the hatchery. Nests moved to the hatchery were buried 30 cm apart in artificially dug nests (40 cm deep). Nests left in place were excavated to count the number of eggs and re-buried. Some nests left in place were also protected using artificial nest covers (see original paper for details). All nests were temporarily covered prior to hatching to enable hatchlings to be counted. After emergence, all nests were dug up and unhatched eggs counted.

A replicated, randomized study in 2004 on a sandy beach in Terengganu, Peninsular Malaysia (15a) found that green turtle *Chelonia mydas* hatchlings from hatcheries released immediately after emerging moved almost twice as fast and had better body condition than hatchlings that were held for 3–6 hours prior to release. Green turtle hatchling running speeds were higher when released immediately (0.12 m/s) compared to when they were held in the hatchery for 1 h (0.11 m/s), 3 h (0.8 m/s), or 6 h (0.7 m/s) before release. Hatchling body condition was similar for newly emerged hatchlings (4.67 g/mm) and hatchlings held for 1 h (4.66 g/mm), but lower for those held for 3 h (4.61 g/mm) or 6 h (4.55 g/mm). In July–October 2004, two hundred hatchlings from 10 hatchery nests (20 hatchlings/nest) were measured for running speed (time to run over a 1.6 m plastic gutter lined with sand, repeated three times/hatchling; see original paper for details) and body condition (ratio of hatchling mass to body length). Hatchlings were measured immediately following emergence or at 1 h, 3 h and 6 h following emergence (5 hatchlings/nest/time held).

A replicated study in 2004 on a sandy beach in Terengganu, Peninsular Malaysia (15b) found that green turtle *Chelonia mydas* hatchlings excavated from hatchery nests immediately after most hatchlings in the nest emerged moved faster and had higher body condition than hatchlings excavated five days after most hatchlings emerged. Hatchlings excavated from nests immediately after the main clutch emergence (0.10 m/s) were faster than hatchlings excavated five days later (0.60 m/s) and had similar running speeds compared to hatchlings that emerged naturally within five days of the main emergence (0.11 m/s). Hatchlings that emerged with the main emergence were the fastest (0.12 m/s). Body condition of hatchlings excavated immediately (4.73 g/mm) was greater than hatchlings that were excavated five days later (4.39 g/mm) or that emerged naturally within five days (4.60 g/mm), and was similar to hatchlings from the main emergence (4.70 g/mm). In July–October 2004, hatchling running speed (time to run over a 1.6 m plastic gutter lined with sand, repeated three times/hatchling; see original paper for details) and body condition (ratio of hatchling mass to body length) was compared between hatchlings excavated immediately after the main emergence (124 live hatchlings from 19 nests); hatchlings excavated five days after main emergence (56 live hatchlings from 13 nests); hatchlings that emerged naturally within five days of the main emergence (16 hatchlings from 6 nests); and hatchlings from the main emergence (200 hatchlings, number of nests not provided). Up to 10 hatchlings were measured/category/nest.

A controlled, before-and-after study in 2005–2012 on a beach in Costa Rica (16) found that relocating olive ridley turtle *Lepidochelys olivacea* nests to an on-beach hatchery with 24-hour monitoring resulted in similar hatching rates to nests that were left in situ but camouflaged. Results were not statistically tested.

The emergence rate of hatchlings from hatchery nests was 77%, compared to 71% of hatchlings from camouflaged in situ nests. The authors reported that egg poaching reduced from 85% in 2005 to 10% of eggs in 2005–2012. Nesting activity was monitored by nightly beach patrols (four 4 h long patrols) in July or August–December in 2006–2012 (98–177 nests laid/year). Nests were either relocated to an on-beach hatchery (363 nests, 40% of total), or camouflaged and left in situ (595 nests, 61% of total) to discourage illegal collecting. Relocated nests were randomly allocated a 1 m^2 plot in the hatchery and dug into the sand. The hatchery was monitored 24 hours a day during the nesting season. Hatchlings were monitored on emergence and nests were excavated after hatching due dates to check hatching success.

A controlled study in 2008 on a sandy beach in Boa Viste, Cape Verde (17) found that loggerhead turtle *Caretta caretta* nests relocated to beach hatcheries without ghost crabs *Ocypode cursor* had higher hatching success and lower predation rates compared to nests left in situ. Hatching success was higher in nests that were relocated to hatcheries (65% success) compared to nests that were left in situ (33% success). Ghost crab predation rates were lower in nests that were relocated to hatcheries (2%) compared to those that were left in situ (55%). Turtle nests were excavated to count the eggs and reburied in either a hatchery (20 nests) or in the same place without any protection (20 nests). Nests were monitored daily until emergence. Hatchlings were counted and released from hatchery on emergence. Hatchling tracks were counted from other nests. All nests were excavated after last emergence and remaining eggs counted for analysis.

A study in 1996–2011 on a sandy beach in Costa Rica (18) found that the majority of olive ridley turtle *Lepidochelys olivacea* eggs relocated to a fenced on-beach hatchery hatched and most hatchlings made it to the sea. Of 1,703 olive ridley turtle nests relocated to a beach hatchery, 78% of eggs hatched (120,015 eggs) and 22% did not (33,986). Of the eggs that hatched, 77% (117,886) emerged successfully and made it to water and 1% (2,129) died. Of nests left in situ, 8 were predated and 566 experienced egg looting. In July–December 1996–2011, nesting activity was monitored by nightly beach patrols (two 3 h patrols/night, 2,401 nights and 2,535 successful nesting events). Turtles were individually marked when encountered (1,239 olive ridleys) and 1,703 (67%) nests (154,001 eggs) were moved to an on-beach hatchery on the beach behind the tide line within 6 h of being laid. The remaining nests (832, 33%) were left in situ. The hatchery was protected by a 2 m high fence buried 40 cm into the sand. Sand in the hatchery was replaced annually. Nests were excavated after hatching due dates to check hatching success.

A randomized study in 2012–2013 on sandy beaches on the pacific coast of Mexico (19) found that relocating olive ridley turtle *Lepidochelys olivacea* nests to an on-beach hatchery resulted in similar hatching success compared to artificially incubating eggs in polystyrene boxes, although turtles from the hatchery had longer shells and faster crawl speeds. Hatching success was similar for nests in the hatchery (82%) and for those incubated in polystyrene-boxes (89% success). Hatchlings relocated to an on-beach hatchery had longer shells (41 mm straight carapace length) and faster crawl speeds (0.018 m/second) than those incubated in polystyrene boxes (length: 39 mm; crawl speed: 0.011 m/second). Carapace width, hatchling weight and righting response time were similar between hatchlings from the hatchery and polystyrene boxes (see original paper for details). In 2012–2013, eggs from 49 nests were moved to one of two treatments: buried in an on-beach hatchery (33 nests) or embedded in sand in polystyrene boxes (16 nests). Upon emergence and movement, ten hatchlings (489 individuals in total) were randomly chosen from each nest to measure size, weight, crawl speed and righting response.

A replicated, randomized, controlled, before-and-after study in 2013 on a sandy beach in Michoacán, Mexico (20) found that olive ridley turtle *Lepidochelys olivacea* hatchlings incubated in an on-beach hatchery weighed less and had elevated stress hormone levels on emergence compared to hatchlings from natural nests. Hatchery hatchlings weighed less (16 g) than natural nest hatchlings (17 g), although other measures of body size such as body length were similar between hatchlings (see paper for details). On emergence, hatchery hatchlings had higher stress hormone levels (corticosterone serum: 31 ng/mL) compared to natural nest hatchlings (27 ng/mL). On arrival at sea, hatchery hatchlings stress hormone levels did not increase compared to the levels at emergence (at sea: 32 ng/mL; emergence: 31 ng/mL), whereas natural nest hatchlings stress hormone levels did increase (at sea: 33 ng/mL; emergence: 27 ng/mL). In 2013, olive ridley turtle nests were relocated to an on-beach hatchery and reburied. Natural nests were located near the hatchery. Seventeen hatchlings each from three hatchery and three natural nests were captured randomly on emergence for to measure size and levels of stress hormones (corticosterone serum) (see original paper for details). A further 10 hatchlings from two hatchery nests and 18 from three natural nests were sampled for stress hormone levels on arrival at sea. These hatchlings were taken to a location 20 m from the sea and set free, and hormone levels were measured when they arrived at the sea.

A study in 2011 on a sandy beach in Mauritius (21) found that moving a green turtle *Chelonia mydas* nest to an on-beach fenced enclosure resulted in most eggs hatching and hatchlings reaching the sea. In total, 26 turtles hatched from 36 eggs, of which 23 were released and reached the sea (three hatchlings were predated by ghost crabs *Ocypode cursor*). The nest was located as part of a survey and eggs were placed into a 3 x 3 m fenced enclosure in the same formation as they had been found in.

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Tortoises, terrapins, side-necked & softshell turtles

- **Two studies** evaluated the effects on tortoise, terrapin, side-necked and softshell turtle populations of relocating nests/eggs to a hatchery. One study was in Costa Rica[1] and Venezuela[2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Reproductive success (2 studies):** One replicated, controlled study in Venezuela[2] found that yellow-headed sideneck turtle eggs relocated to a hatchery had higher hatching success than both natural nests and artificially incubated eggs. One study in Costa Rica[1] reported that 80% of Nicaraguan slider eggs in a hatchery hatched successfully.

BEHAVIOUR (0 STUDIES)

A study in 1991 on a river in northern Costa Rica (1) reported that some Nicaraguan slider *Trachemys emolli* eggs taken from the wild to a hatchery (as part of a ranching program) hatched successfully. The author reported that approximately 80% of eggs collected hatched successfully in the hatchery, and that 30% of hatchlings were released into the wild. In 1991, eggs from 310 nests were collected (average of 20 eggs/nest) within 24 hours of laying and reburied in soil in an enclosed area. Collection was carried out by local people, who received 50% of the funds generated by sale of the turtles.

A replicated, controlled study in 2009 on two rivers in Southern Venezuela (2) found that relocating eggs of yellow-headed sideneck turtles *Podocnemis unifilis* to a hatchery resulted in higher hatching success compared to eggs from natural nests and eggs incubated artificially. Results were not statistically tested. Hatching success was higher for eggs from the hatchery (88%) than for eggs from both natural nests (63%) and artificially incubated eggs (42%). Five eggs each from 27 nests (136 total) at one river were moved to a hatchery and reburied in a trench (200 x 40 x 30 cm) using sand from the nesting site. The area was protected by a 1.5 m metal mesh fence, and two staff monitored the site and poured 5 litres of water over the trench each week. All eggs from 13 nests (401 total) at the second river were placed in sand-filled polystyrene containers and incubated indoors in ambient conditions. All eggs from a further 51 nests from the first river were left in place. In February 2009, a 6 km and 13 km stretch of each river was searched for nests. In May, these locations were revisited to assess hatching success.

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Snakes & lizards

- We found no studies that evaluated the effects of relocating nests/eggs to a hatchery on snake and lizard populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Crocodilians

- We found no studies that evaluated the effects of relocating nests/eggs to a hatchery on crocodilian populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Tuatara

- We found no studies that evaluated the effects of relocating nests/eggs to a hatchery on tuatara populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

14.23. Relocate nests/eggs for artificial incubation

Background

Relocating eggs to artificially incubate them may be used as a way to maximise hatching success as the eggs will no longer be vulnerable to natural threats such as predation. Consideration must be given to the potential impacts of incubation conditions (for example temperature and humidity) on the sex, size, shape, colour, behaviour, movement ability and post-hatching growth of reptile hatchlings (Warner & Andrews 2002, Booth *et al*. 2006).

This action includes studies where eggs are incubated in artificial conditions, which ranges from controlled laboratory settings to using polystyrene boxes to incubate eggs in buildings. Studies are also included where gravid females are brought into captivity to lay eggs, but eggs are taken away from the females for artificial incubation.

Due to the number of studies found, this action has been split by species group, though no studies were found for amphisbaenians.

For studies that discuss the effects of relocating and re-burying eggs in natural habitats to avoid threats, see *Relocate nests/eggs to a hatchery* and *Relocate nests/eggs to a nearby natural setting (not including hatcheries)*.

See also *Maintain wild-caught, gravid females in captivity during gestation* and *Alter incubation temperatures to achieve optimal/desired sex ratio*.

Booth D.T. (2006) Influence of incubation temperature on hatchling phenotype in reptiles. P*hysiological and Biochemical Zoology*, 79, 274–281.

Warner D.A. & Andrews R.M. (2002) Laboratory and field experiments identify sources of variation in phenotypes and survival of hatchling lizards. *Biological Journal of the Linnean Society*, 76, 105–124.

Sea turtles

- **Fifteen studies** evaluated the effects of relocating nests/eggs for artificial incubation on sea turtle populations. Three studies were in Suriname[2,4,5] and the USA[6,9,13], two were in each of Costa Rica[1,15], Malaysia[7,8], the Dominican Republic[10,12] and Mexico[11,14] and one was in the Cayman Islands[3].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (12 STUDIES)

- **Reproductive success (12 studies):** One of four controlled studies (including three replicated studies) in Suriname[5], the Dominican Republic[10,12], the USA[13] found that kemp's ridley nests relocated for artificial incubation had higher hatching success than natural nests[13]. One of the studies[12] found that leatherback turtle nests relocated for artificial incubation had lower hatching success than natural nests. One of the studies[10] found that hawksbill turtle nests relocated for artificial incubation had similar hatching success compared to natural nests. The other study[5] found that hatching success of leatherback and green turtle nests relocated for artificially incubation was similar to natural nests above the high tide line and may have been higher than for natural nests washed over by sea swells. This study[5] also found higher embryo mortality in artificially incubated nests compared to natural nests. Three studies (including one randomized, controlled study) in the USA[6], Mexico[14] and Malaysia[7] found that loggerhead[6], olive ridley[14] and leatherback turtle[7] nests relocated for artificial incubation had similar hatching success compared to nests relocated to an on-beach hatchery. One study[7] also found that careful handling of eggs during the first five days of incubation did not affect hatching success. Four studies (including one replicated study) in Surinam, Ascension Island and Costa Rica[1], the Cayman Islands[3], the USA[9] and Mexico[11] reported that hatching success of green[1,3], loggerhead[9] and olive ridley turtle[11] nests relocated for artificial incubation varied from 26% to >90%. One study[3] also reported that hatching success from two trials was 30% and 58% in foam-packed boxes and 26% and 48% in sand-packed boxes. One study[11] also reported that hatching success was 60–89% in 14 of 18 years. One replicated, randomized, controlled study in Costa Rica[15] found that olive ridley turtle eggs artificially incubated in low oxygen conditions had lower hatching success than those in normal oxygen conditions.
- **Condition (2 studies):** One replicated, controlled study in Suriname[5] found that leatherback and green turtle nests relocated for artificial incubation had more instances of embryo deformities than natural nests. One randomized, controlled study in Mexico[14] found that relocating olive ridley nests for artificial incubation had mixed effects on hatchling size and movement compared to those relocated to an on-beach hatchery.

BEHAVIOUR (0 STUDIES)

OTHER (3 STUDIES)

- **Offspring sex ratio (3 studies):** Three replicated studies (including two controlled studies) in Suriname[2,4] and Malaysia[8] found that green turtle[2] and leatherback turtle[4,8] nests relocated for artificial incubation produced fewer female hatchlings than eggs from natural nests[2,4] and/or that all sexed hatchlings that were artificially incubated were male[4,8].

A replicated study in 1971–1973 of turtle eggs collected from Surinam, Ascension Island and Costa Rica (1) found that at least half of green turtle *Chelonia mydas* eggs that were relocated and artificially incubated hatched successfully over three years. In the first year, 14,346 of 30,000 (48%) green turtle eggs hatched successfully. In the second year, 34,527 of 61,257 (56%) green turtle eggs hatched successfully. In the third year, 76,024 of 97,312 (78%) green turtle eggs hatched successfully. In 1971–1973, seven batches of green turtle eggs (14,803–63,404 eggs/batch, 1–3 batches/year) were collected from Surinam (3 batches), Ascension Island (2 batches) and Costa Rica (2 batches), placed in Styrofoam incubation boxes with sand (approximately 88 eggs/box) and relocated to open-sided wooden shelters at a turtle farm on Grand Cayman. The top layer of sand inside the boxes was periodically moistened with water and was removed 3–5 days before hatching was expected. Numbers of infertile eggs, unviable and viable hatchlings were recorded after hatchlings had emerged.

A replicated, controlled study on a sandy beach in Suriname (2) found that artificially incubated green turtle *Chelonia mydas* eggs produced a lower percentage of female hatchlings compared to natural nests. The percentage of female hatchlings was lower in nest boxes (41 of 97, 42% [numbers from table]) compared to natural nests (77 of 120, 64%). Temperatures in incubation boxes may have been cooler than those in the sand of the nesting beach (box: 27.4°C; beach: 28.8°C), and average incubation periods were longer (box: 63 days; beach: 57 days), although these results were not tested statistically. Ten clutches of eggs were incubated in polystyrene boxes with around 88 eggs/box, and a further 12 clutches were left in the sand where they were laid. Temperature was measured in an additional incubation box, though the hatchlings from this box were not included in the analysis. Temperatures on the beach were taken at a depth of 80 cm, in an unshaded area where turtles nest. Temperature readings were taken at 3 h intervals in early June. Ten hatchlings from each clutch were euthanized and their gonads were dissected to determine the sex.

A study in 1979 at a sea turtle farm in the Cayman Islands (3) found that 26–58% of green turtle *Chelonia mydas* eggs collected from a captive colony hatched when artificially incubated in sand-packed and foam-layered Styrofoam boxes. Results were not statistically tested. In a small-scale trial, hatching success of eggs incubated in foam-packed boxes was 58% (undeveloped eggs: 15%, developing eggs that didn't hatch: 27%) and hatching success of eggs incubated in sand-packed boxes was 48% (undeveloped eggs: 20%, developing eggs that didn't hatch: 31%; egg numbers not provided). In a larger trial, hatching success of eggs incubated in foam-packed boxes was 30% (1,311 of 4,400 eggs) compared to 26% hatching success of eggs incubated in sand-packed boxes (11,004 of 42,000 eggs). In an initial trial to compare incubation approaches, nine green turtle clutches were divided, and half of the eggs were placed in Styrofoam boxes packed in sand and the other half were placed in Styrofoam boxes packed in between layers of perforated polyethylene foam (3.8 cm thick). Each box contained 56–97 eggs. Following this trial 4,400 eggs collected in 1979 were incubated in foam-packed boxes and hatching success compared to 42,000 eggs incubated in sand-packed boxes.

A replicated, randomized, controlled study in 1980 and 1982 on a sandy beach in Suriname (4) found that artificially incubating leatherback turtle *Dermochelys coriacea* nests in Styrofoam boxes produced all male hatchlings whereas natural nests produced mixed sex ratios and reburied nests produced all female hatchlings. Leatherback turtle eggs incubated in Styrofoam boxes produced no female hatchlings, compared to 30–100% of female hatchlings in natural nests and 100% female hatchlings in reburied nests. Incubation duration was 70–73 days in

Styrofoam boxes and 60–66 days in natural nests (results not statistically tested). Leatherback turtle eggs from five clutches laid in 1980 and 10 clutches laid in 1982 were incubated in Styrofoam boxes (45–60 eggs/box). In 1980, ten embryos were sampled and sexed prior to hatching. In 1982, ten hatchlings were randomly selected from each box after emergence, euthanised and sexed. Sex ratios were compared to 10 hatchlings/clutch of two naturally-incubated nests laid in 1980, six naturally-incubated nests laid in 1982 and two clutches laid in 1982 below the tide line that were reburied elsewhere on the beach.

A replicated, controlled study in 1982 on a sandy beach in Suriname (5) found that leatherback *Dermochelys coriacea* and green *Chelonia mydas* turtle nests incubated in Styrofoam boxes had comparable hatching success to natural nests not washed over by sea swells, but greater incidences of embryonic mortality and deformity than natural nests. Average hatching success of turtle clutches relocated to Styrofoam boxes was 60–73% compared to 33–67% in natural nests washed by sea swells, and 62–82% in natural nests not washed by sea swells (results were not statistically tested). Embryonic mortality and deformity occurred more often in eggs incubated in Styrofoam boxes (mortality: 26–33% of eggs, deformity occurred in 50–88% of clutches) than natural nests (mortality: 8–21% of eggs, deformity: 10–20% of clutches). No eggs incubated in Styrofoam boxes were predated whereas in natural nests 17–27% of leatherback and 11–12% of green turtle eggs were predated. Some leatherback turtle nests were reburied further up the nesting beach for comparison (see original paper, or "Relocate eggs/nests away from threats"). Nesting turtles were surveyed at least once/week in March – August 1982 on a 12 km long beach. Nests laid below the spring high tide line were relocated the next day to a hatchery for incubation above ground in Styrofoam boxes (11 leatherback clutches, 45 eggs/box; 8 green turtle clutches, 88 eggs/box). Relocated and natural nests (~35 green turtle and ~30 leatherback nests) were excavated after emergence to evaluate hatching success.

A replicated study in 1983–1984 on two sandy beaches in Florida and Georgia, USA (6) found that loggerhead turtle *Caretta caretta* nests relocated to incubators had similar hatching success compared to eggs reburied in an on-beach hatchery. Hatching success was similar for artificially incubated eggs (135 of 163, 83% of eggs hatched) and eggs relocated to a hatchery (3,608 of 5,100, 71% of eggs hatched). An additional five nests from another beach had similar hatching success (543 of 588, 92% hatched from 5 nests) (result was not statistically tested). Nine of 50 relocated clutches (18%) were partially destroyed by ghost crab *Ocypode quadrata* predation, cold weather or drifting sand. In 1983, all loggerhead nests on one beach (53 nests) were relocated due to risk of total failure (predators, storm tides, poachers). Three clutches were placed in glass-fronted polystyrene incubators (38 x 38 x 19 cm) and 50 clutches were reburied in hand-dug nests in a fenced area on a nearby dune. In 1984, five nests from a second beach were relocated for artificial incubation. Hatching success was assessed following emergence of hatchlings.

A replicated, randomized study in 1986 on one sandy beach in Rantau Abang, Malaysia (7; same experimental set-up as 8) found that relocating leatherback turtle *Dermochelys coriacea* eggs for artificial incubation in Styrofoam boxes resulted in similar hatching success compared to eggs that were relocated to an on-beach hatchery. Hatching success was similar for eggs from Styrofoam boxes

(52–100%) and eggs from the hatchery (13–92%). In addition, careful handling of eggs during the first five days of incubation did not affect hatching success (handled eggs: 70–100%; non-handled eggs: 52–100%). Eggs were collected from four natural nests (only yolked eggs of normal size) and four groups of 23–25 eggs each were incubated in Styrofoam boxes with egg handling during the first five days; Styrofoam boxes with no handling; or in an on-beach hatchery (98 eggs/treatment). Eggs in the on-beach hatchery were buried 60 cm deep, and the nests were surrounded with chicken mesh after 50 days to capture emerging hatchlings. Half on the Styrofoam boxes were kept in a well-ventilated shed, and the others were kept in an enclosed laboratory. Hatching success was measured by counting the number of hatchlings that emerged.

A replicated, randomized study in 1986 on one sandy beach in Rantau Abang, Malaysia (8; same experimental set-up as 7) found that relocating leatherback turtle *Dermochelys coriacea* eggs for artificial incubation in Styrofoam boxes resulted in all male hatchlings. Of the hatchings that were sexed, 29 of 29 were male. Eggs were collected from natural nests (only yolked eggs of normal size) and three groups of 25 eggs were incubated in Styrofoam boxes (temperature range 27–29°C). Temperatures were monitored three times/day (0900, 1200 and 1500 h) with a mercury thermometer inserted horizontally through a hole in the side of the box. A sample of 9, 9 and 11 hatchlings from each box were selected for sexing. These hatchlings were euthanised in chloroform and sex was determined by removing and examining the gonads.

A study in 1995–1996 on a sandy beach on the southeastern coastline in Virginia, USA (9) found that most loggerhead turtle *Caretta caretta* eggs artificially incubated in plastic planter pots with predator proof cages hatched. Hatchling success of three loggerhead turtle nests artificially incubated in plastic planter plots was 94%, 88% and 42% (numbers of eggs not provided). In 1995–1996, three late loggerhead turtle nests were relocated with their nesting chamber sand to large plastic tree planter pots (depth: 67 cm, top diameter: 76 cm) lined with burlap and damp sand (see original paper for details). Nests were covered with predator-proof nest cages and placed in enclosed heated building. Hatchlings were released on the natal nesting site for imprinting and released in the ocean.

A replicated, controlled study in 2007–2010 on 12 sandy beaches in Saona Island, Dominican Republic (10) found that artificially incubating hawksbill turtle *Eretmochelys imbricata* nests in boxes had similar hatching and emergence success compared to nests left in situ. Artificially incubated hawksbill turtle nests had similar hatching success (72–81%) and emergence success (69–80%) compared to nests left in situ (hatching success: 72–78%; emergence success: 67–72%). In 2007–2010, hawksbill turtle nesting activity was monitored on 12 beaches (0.01–2.10 km long) and nests deemed vulnerable to predation or harvesting were removed for artificial incubation in plastic boxes filled with sand and polyurethane foam (see original paper for details). Artificial incubation boxes were placed in a facility near one of the beaches (4 m long x 3 m wide) with a sand floor and wire mesh and corrugated metal walls. Hatching and emergence success was determined for clutches that were artificially incubated (20–41 nests/year, 119 total nests) and left in situ (7–21 nests/year, 49 total nests).

A study in 1993–2010 on four sandy beaches in a single bay in Nayarit, Mexico (11) found that at least half of artificially incubated olive ridley turtle *Lepidochelys olivacea* nests hatched successfully each year. Over 18 seasons of artificially incubating olive ridley turtle nests, hatching success was 50–59% in two years, 60–69% in five years, 70–79% in three years, 80–89% in six years and >90% in two years. Number of hatchlings released varied between 2,555 in 1997 and 23,467 in 2006. Four turtle nesting beaches (2–8 km long) were monitored during the peak nesting season (July–November) for two nights/week in 1993–1999 and seven nights/week in 2000–2010. Nests were collected (1.4 nests/day) and artificially incubated in polystyrene boxes (40 x 30 x 50 cm, wall thickness: 2 cm; see original paper for details) in an indoor facility on one of the beaches. Hatching success was evaluated once 20 hatchlings had emerged by calculating the proportion of live and dead neonates.

A replicated, controlled study in 2006–2010 on five sandy beaches in southwest Dominican Republic (12) found that leatherback turtle *Dermochelys coriacea* nests relocated for artificial incubation tended to have lower hatching rates than nests left in situ on a beach that was patrolled by park rangers. Results were not statistically tested. Over two years, nests relocated for artificial incubation tended to have lower hatching success (east hatchery: 51–58%; west hatchery: 34–43%) than nests left in situ (74–85%). In March–August 2008–2009, nests were relocated from three beaches in the east (35 nests) and two in the west (31 nests) of a national park (1,374 km^2). On western beaches, which had limited human access, 43 nests were left in place and monitored to hatching. Eggs from relocated nests were placed polystyrene boxes with sand and moved to nearby hatcheries (one in the east, one in the west, enclosed wooden barracks with concrete floor and metal roof). On the western beaches where nests were left in situ, nightly patrols were carried out by government rangers 2–3 nights/week in April–May.

A controlled study in 1979–2014 on sandy beaches in the Gulf of Mexico, Texas, USA (13) found that artificially incubated kemp's ridley turtle *Lepidochelys kempii* nests had higher hatching success than nests left in situ. Results were not statistically tested. Emergence success of artificially incubated kemp's ridley turtle nests was 82% and hatching success of in situ nests was 62%. The authors reported that many in situ nest hatchlings did not make it to the sea successfully. Over the 37-year programme, 130,847 artificially incubated hatchlings emerged successfully and were released. In 1979–2014, the majority of kemp's ridley turtle nests laid in the USA were collected for artificial incubation (1,606 nests) and a small number hatched in situ (61 nests). Hatching rates were assessed for 26 in situ nests laid in 1979–2008.

A randomized, controlled study in 2012–2013 on sandy beaches on the pacific coast of Mexico (14) found that relocating olive ridley turtle *Lepidochelys olivacea* nests for artificial incubation resulted in similar hatching success, but mixed effects on hatchling size and behaviours, compared to nests moved to on-beach hatcheries. Hatching success was similar for eggs artificially incubated (89% success) and placed in beach hatcheries (82% success). Polystyrene-box incubated hatchlings had smaller straight carapace length (39 mm) and slower crawl speeds (0.01 m/second) than those from beach hatcheries (length: 41 mm; crawl speed: 0.02 m/second). Carapace width, hatchling weight and righting response time were similar between polystyrene box and beach hatchery nests (see original paper for details). Eggs from 49 nests were moved to one of two treatments: embedded in sand in polystyrene boxes (16) or buried in an on-beach hatchery (33). Upon emergence and movement, 10 hatchlings were randomly chosen from each nest (489 individuals in total) for measuring, weighing and to take part in fitness tests.

A replicated, randomized, controlled study in 2015 in laboratory conditions in Costa Rica (15) found that olive ridley turtle *Lepidochelys olivacea* eggs artificially incubated in normal oxygen conditions had better hatching success, but were more vulnerable to being inverted, than eggs initially artificially incubated in low oxygen ('hypoxic') conditions. Hatching success of olive ridley turtle eggs initially incubated in any one of three hypoxic conditions was lower (Perspex box with nitrogen: 23 of 75 eggs; zip lock bag with nitrogen: 14 of 71 eggs; vacuum-sealed plastic bag: 34 of 79 eggs) than eggs incubated in normal oxygen conditions (53 of 78 eggs). Hatching success in hypoxic-maintained eggs was similar whether or not eggs were inverted during the incubation process, whereas when eggs were incubated in normal oxygen conditions, inverting eggs lowered hatching success (see original paper for details). For three days after collection, olive ridley eggs collected from six nesting females in October–November 2015 were either kept in normal oxygen conditions in a sand-filled incubator (78 eggs), or in one of three 'hypoxic' containers: a Perspex box filled with nitrogen (75 eggs), a plastic bag filled with nitrogen (71 eggs), or a vacuum-sealed bag (79 eggs; 13–24 eggs/hypoxic container, four containers/type). A subset of eggs from each treatment (normal oxygen: 10 eggs; Perspex box: 10 eggs; zip lock bag: 7 eggs; vacuum-sealed bag: 10 eggs) were inverted 180° horizontally after three days and compared to equivalent numbers of eggs/treatment that were not inverted. After experimental treatments, eggs were either buried in a hatchery or maintained in incubators and hatchlings were counted on emergence.

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Tortoises, terrapins, side-necked & softshell turtles

- **Seventeen studies** evaluated the effects of relocating nests/eggs for artificial incubation on tortoise, terrapin, side-necked & softshell turtle populations. Ten studies were in the USA[2-4,6-8,13-16], two were in each of the Galápagos[1a,1b] and China[5,11] and one was in each of Brazil[9], Venezuela[10] and Thailand[12].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (17 STUDIES)

- **Reproductive success (16 studies):** Two of three replicated controlled studies (including one randomized study) in Brazil[9], Venezuela[10] and the USA[15] found that Hilaire's side-necked turtle[9] and bog turtle[15] nests relocated for artificial incubation had higher hatching success, or likely had higher success[9], than natural nests[9,15].The other study[10] found that yellow-headed sideneck turtle nests relocated for artificial incubation had lower hatching success than natural nests and nests moved to an on-beach hatchery. One replicated study in the Galápagos[1a] reported that hatching success of five subspecies of giant tortoise nests relocated for artificial incubation was 35–100%, compared to 76–85% for natural nests of two sub species. Six of eight studies (including four replicated studies) in the USA[2,3,7,8,13,14,16] and China[11] reported that hatching success for artificially incubated eggs, including eggs recovered from road-killed turtles[3], was 60–97%[2,3,7,14,16], or that 314 hatchlings emerged, and 14 eggs did not hatch[8]. One study[7] also found that eggs collected from the wild had similar hatching success compared to oxytocin-induced eggs. The other two studies[11,13] reported that hatching success of eggs or clutches was 39–54%. One replicated study in the Galápagos[1b] reported that hatching success of giant tortoise nests relocated for artificial incubation may have been higher for nests relocated longer after laying. One replicated study in the USA[4] found that high levels of CO_2 during artificial incubation of pond slider and Mississippi map turtle eggs resulted in lower hatching success compared to low CO_2 levels. One replicated, randomized study in China[5] found that hatching success of artificially incubated Chinese three-keeled pond turtle eggs was similar across all temperatures tested. One randomized study in the USA[6] found that hatching success of artificially incubated snapping turtle eggs was highest at intermediate levels of soil moisture.

- **Survival (3 studies):** Two studies (including one replicated study) in the USA[8,13] reported that after relocating smooth softshell turtle[8] and gopher tortoise[13] nests for artificial incubation, two of 314[8] and three of 36[13] hatchlings died soon after emergence. One randomized study in the USA[6] found that survival of artificially incubated snapping turtle hatchlings was lower at high soil moisture levels compared to intermediate moisture levels.
- **Condition (4 studies):** One replicated, randomized, controlled study in Brazil[9] found that Hilaire's side-necked turtle nests relocated for artificial incubation produced heavier hatchlings that were larger in four of five measures compared to hatchlings from natural nests. Two replicated studies (including one randomized study) in China[5,11] found that modifying incubation temperatures of Chinese three-keeled pond turtle[5] or Asian yellow pond turtle[11] eggs had mixed effects on hatchling size and mobility[5] or different effects on growth depending on the population eggs were sourced from[11]. One replicated study in Thailand[12] found that artificially incubating snail-eating turtle eggs at higher temperatures resulted in more embryos with physical deformities.

BEHAVIOUR (0 STUDIES)

OTHER (1 STUDY)

- **Offspring sex ratio (1 study):** One replicated study in the USA[4] found that high levels of CO_2 during artificial incubation of pond slider and Mississippi map turtle eggs resulted in a lower proportion of male hatchlings compared to low CO_2 levels.

A replicated study in 1966–1971 in an artificial incubating facility in Galápagos, Ecuador (1a) found that eggs from five subspecies of Galápagos giant tortoise hatched successfully after artificial incubation. Results were not statistically tested. Hatching success of artificially incubated *Geochelone elephantopus ephippium* eggs was 51% (158 of 312 eggs), *G. e. darwini* eggs was 37% (44 of 118 eggs hatched, 83 embryos died, 71 infertile eggs), *G. e. hoodensis* eggs was 63% (20 of 32 eggs hatched, 28 embryos died, 46 infertile eggs), *G. e. porteri* eggs was 35% (6 of 17 eggs hatched, 11 eggs infertile), and *G. e. chathamensis* eggs was 100% (3 of 3 eggs hatched). Hatching success of *G. e. ephippium* and *G. e. porteri* eggs incubated in undisturbed natural nests in the wild was 82% (103 of 133 eggs hatched, 4 embyros died, 18 infertile eggs) and 76% (391 of 520 eggs hatched, 21 embryos died, 101 infertile eggs) respectively. In the 1969/1970–1970/1971 nesting seasons, giant tortoise eggs laid by *G. e. ephippium* (312 total eggs), *G. e. darwini* (118 total eggs), *G. e. hoodensis* (32 total eggs), *G. e. porteri* (17 total eggs) and *G. e. chathamensis* (3 total eggs) were transported to an artificial incubation facility (1–2 hours on foot and 5–6 hours by boat). Eggs were incubated in wooden boxes with a soil substrate (see original paper). Hatching success and egg fertility was compared between subspecies and to naturally incubated, undisturbed *G. e. ephippium* and *G. e. porteri* eggs laid in the same seasons.

A replicated study in 1969–1971 in an artificial incubating facility in Galápagos, Ecuador (1b) reported that Galápagos giant tortoise *Geochelone elephantopus ephippium* egg hatching success was higher when eggs were left longer in natural nests before being relocated for artificial incubation. Results were not statistically tested. Of artificially incubated giant tortoise eggs, 74% (43 of 71) moved at 10–15 weeks, 67% (4 of 6) moved at 7–9 weeks, 19% (5 of 29) moved at 4–6 weeks, and 19% (5 of 27) moved at 0–2 weeks hatched successfully. In undisturbed natural nests, 82% (103 of 133) of *G. e. ephippium* eggs and 76% (391 of 520) *G. e. porteri* eggs hatched. In the 1969/1970 and 1970/1971 nesting seasons, giant tortoise eggs were transported at 10–15 weeks (71 eggs from 16 clutches), 7–9 weeks (6 eggs from 2 clutches), 4–6 weeks (29 eggs from 6 clutches) and 0–2 weeks (27 eggs from 5 clutches) after laying to an artificial incubation facility. Eggs were incubated in wooden boxes with a soil substrate (see original paper). Hatching success was compared to clutches left undisturbed laid by *G. e. ephippium* (133 eggs from 26 clutches) and *G. e. porteri* (520 eggs from 55 clutches) in the same nesting seasons.

A study in 1985–1986 at Columbus Zoo, Ohio, USA (2) found that two of three incubated eggs laid in captivity by a wild-caught gravid gibba turtle *Mesoclemmys gibba* hatched successfully. A female laid three eggs over a one-month period, and two hatched successfully after 154 and 164 days of incubation. The third egg failed during incubation. A wild-caught female was acquired in 1985 and housed along with a range of other turtle species in a 140 cm square display tank, with 50 cm deep water and a basking spot. Water temperature was 20–24°C and air temperatures were 24–32°C. Eggs were incubated at 26–31°C in sealed 1 gallon jars in a 1:1 mixture of vermiculite and water (by weight), and jars were vented every 4–6 weeks.

A replicated study in a laboratory in Illinois, USA (3) found that after incubating eggs recovered from road-killed red-eared sliders *Trachemys scripta elegans*, more than half of the eggs hatched successfully, and hatching success was higher for eggs from turtles found with intact carapaces compared to those with open carapaces. Forty-three of 67 (64%) eggs hatched successfully. Hatching success was higher for eggs recovered from turtles with intact shells (30 of 35, 86 %) compared to those with shells that had been opened (13 of 32, 41 %). Of 32 turtles that were found on a road having been hit by a vehicle, nine contained 2–21 unbroken eggs. One turtle survived and was later released after laying eggs. Unbroken eggs were transferred to a laboratory and partially buried in perlite incubation medium in plastic containers (32 x 19 x 10 cm), with aluminium foil layered under the lid. Clutches were incubated separately. A road was searched for turtles hit by vehicles at least twice daily during the nesting season (months not given).

A replicated study in 1992 in Pennsylvania, USA (4) found that most artificially incubated pond slider *Trachemys scripta* and Mississippi map turtle *Graptemys pseudogeographica kohni* eggs hatched successfully, but higher CO_2 concentrations during incubation led to lower hatching success for pond sliders in two of three comparisons and a higher proportion of female hatchlings for both species. Compared to 0% CO_2, hatching success was lower at the highest concentration of CO_2 for pond sliders (24 of 25 eggs at 0% CO_2 vs 4 of 25 at 15% CO_2; all with 21% oxygen), but remained similar at medium CO_2 concentrations for both species (pond sliders: 21 of 25 at 10% CO_2; map turtles: 12 of 14 at 0% CO_2 vs 7 of 14 at 10% CO_2; all with 21% oxygen). The proportion of males to females was lower at higher concentrations of CO_2 for pond sliders (0%: 22:2; 10%: 11:12; 15%: 4:6) and map turtles (0%: 10:2; 10%: 3:6). A separate trial for pond slider eggs at high CO_2 (15%) and 10% oxygen resulted in 0 of 24 eggs hatching successfully and a sex ratio of one male to six female embryos. In 1992, gravid, wild female turtles were injected with oxytocin to obtain their eggs. Groups of 25 pond slider eggs were incubated at 0, 10 or 15% CO_2, and groups of 14 map turtle eggs were incubated at 0 or 10% CO_2, all with 21% oxygen. A further 24 pond slider eggs were assigned to a treatment involving high CO_2 (15%) and reduced oxygen levels (10%).

A replicated, randomized study in 2001 in a laboratory setting in Hangzhou, China (5) found that artificially incubating Chinese three-keeled pond turtle *Chinemys reevesii* eggs at different temperatures did not influence hatching success, but did influence four of five measure of hatchling size and six of 16 comparisons of locomotor performance. Hatching success was similar across all incubation temperatures (73–96%). Four of five measures of hatchling size were affected by temperature (see paper for details). Locomotor performance (four measures of swimming and crawling performance) was better for hatchlings incubated at 27 and 30°C in six of 16 comparisons and similar in the remaining 10 comparisons compared to those incubated at 24 and 33°C. In July 2001, a total of 111 viable eggs (from a private hatchery) were incubated in plastic boxes in moist vermiculite at 24°C (24 eggs), 27°C (28 eggs), 30°C (28 eggs) or 33°C (30 eggs). Crawling and swimming performance of hatchlings was assessed by chasing them along a "racetrack" covered with sand (crawling) or 50 mm of water (swimming).

A randomized study in laboratory conditions in Indiana University, Kokomo, USA (6) found that snapping turtle *Chelydra serpentine* eggs artificial incubated in substrate with higher moisture levels had higher hatching success up to a threshold, after which hatching success was lower. Snapping turtle eggs artificially incubated in 9% and 7% soil moisture had the highest hatching success (7%: 14 of 20 eggs hatched; 9% 16 of 20 eggs hatched) compared to lower soil moisture levels (3%: 9 of 20; 5%: 10 of 20 eggs hatched) or higher soil moisture levels (11%: 9 of 20; 12% 8 of 20 eggs hatched). Survivorship of hatchlings was significantly lower in at 13% moisture level compared to 7% or 9% moisture level (data presented as statistical model outputs). Freshly laid eggs were collected from the wild in June 2002 (120 total eggs from six clutches). Eggs were incubated under one of six soil moisture conditions: 3, 5, 7, 9, 11, and 13% water content (two nest boxes of 10 eggs/moisture level). Eggs were buried to 3 cm depth in the soil and incubated at 25°C.

A replicated study in 1978–2006 in a laboratory in the USA (7) found that when eastern painted turtle *Chrysemys picta picta* eggs from natural nests and from turtles induced with oxytocin were artificially incubated, most hatched successfully. Fifty-seven of 62 (92%) of oxytocin-induced eggs and 58 of 60 (97%) natural nest eggs hatched successfully, and there was no difference in the hatching success or incubation period (average of 58 days) between oxytocin-induced and natural eggs. Sixty painted turtle eggs were collected from eight wild nests, and 14 turtles were collected before laying and induced with oxytocin, yielding 62 eggs. All eggs were incubated in a 50:50 mix by weight of vermiculite and water. Oxytocin (1.4–2.5 units/100 g) was injected into each turtle using a syringe.

A study in 2005–2006 in man-made sandbar habitat in Arkansas, USA (8) found that most smooth softshell turtle *Apalone mutica* nests moved to an outdoor enclosure for incubation hatched. In total, 314 hatchlings emerged from 26 clutches over two nesting seasons. Fourteen eggs from seven clutches did not hatch and two hatchlings died soon after hatching. Turtle eggs were collected from natural nests in May-June 2005–2006 and reburied in an outdoor enclosure in a laboratory facility (2005: 12 eggs each from 10 clutches; 2006: 10–21 eggs each from 16 clutches). The nesting area was monitored twice daily and videoed continuously in the last predicted week prior to hatching.

A replicated, randomized, controlled study in 2004–2005 on one grassy bank of a river delta in Rio Grande do Sul state, Brazil (9) found that artificially incubating Hilaire's side-necked turtle *Phrynops hilarii* eggs resulted in higher hatching success and larger hatchlings compared to eggs incubated in natural nests. Hatching success was higher for artificially incubated eggs (hatching success/nest 50–100%, 25 of 28 eggs hatched) compared to eggs from natural nests (hatching success/nest 43–75%, 30 of 50 eggs hatched), though this result was not tested statistically. Hatchlings from artificially incubated nests were heavier than those from natural nests (artificial: 14 g; natural: 9 g), and larger in four of five measures (see paper for details). In September 2004, six natural turtle nests were selected and 40% of eggs were removed for artificial incubation (28 eggs), and the rest left in place (50 eggs). Natural nests were covered with a plastic screen. Removed eggs were placed in cooler boxes (1,000 x 400 x 350 mm) in moist vermiculite (2:1 ratio with water by volume), and additional water was added whenever the vermiculite was dry. Hatchlings were counted, weighed and released at the nesting site two weeks after hatching.

A replicated, controlled study in 2009 on two rivers in Southern Venezuela (10) found that artificially incubating eggs of yellow-headed sideneck turtles *Podocnemis unifilis* resulted in lower hatching success compared to eggs moved to an on-beach hatchery and eggs from natural nests. Results were not statistically tested. Hatching success was lower for artificially incubated eggs (42%) than for eggs from both the on-beach hatchery (88%) and natural nests (63%). Eggs that were artificially incubated came from locations where all eggs from a further 74 nests had been harvested by people. All eggs from 13 nests (401 total) at one river were placed in sand-filled polystyrene containers and incubated indoors in ambient conditions. Five eggs each from 27 nests (136 total) at the second river were moved to a hatchery and reburied in a trench (200 x 40 x 30 cm) using sand from the nesting site. The area was protected by a 1.5 m metal mesh fence, and two staff monitored the site and poured 5 litres of water over the trench each week. All eggs from a further 51 nests from the second river were left in place. In February 2009, a six and 13 km stretch of each river was searched for nests. In May, these locations were revisited to assess hatching success

A replicated study in 2009 in an artificial setting in Zhejiang, China (11) found that some Asian yellow pond turtle *Mauremys mutica* relocated for artificial incubation hatched successfully, and that higher incubation temperatures resulted in higher growth rates for hatchlings from one of two populations. Overall, 19 of 35 (54%) clutches of artificially incubated eggs hatched successfully (2 eggs/clutch; one incubated at 26°C and one at 30°C). Growth rate of hatchlings sourced from a more southerly population was lower for those incubated at 26°C (0.06 g/day) compared to those incubated at 30°C (0.1 g/day), whereas hatchlings from a more northerly population grew at similar rates at both incubation temperatures (0.04–0.05 g/day). Initial hatchling mass was similar for those incubated at 26°C (5–6 g) and those incubated at 30°C (5–6 g). In 2009, fifteen clutches of eggs were collected from a more southerly wild population (Hainan province) and 20 clutches were collected from a more northerly population (Zhejiang province). Eggs were individually incubated in jars with moist vermiculite, and one egg from each clutch was incubated at 26°C and one was incubated at 30°C. Hatchlings were weighed soon after emergence and then maintained at 28°C for 90 days and weighed again to measure growth rate.

A replicated study in 2011–2012 in Phra Nakhon Si Ayutthaya province, Thailand (12) found that artificially incubating snail-eating turtle *Malayemys macrocephala* eggs at higher temperatures resulted in more physical deformities in embryos. Higher incubation temperatures resulted in more embryos having physical deformities (1% at 26°C; 3% at 29°C; 30% at 32°C). The proportion of infertile eggs did not differ between the temperature treatments (8% at 26°C; 4% at 29°C; 11% at 32°C). In 2011–2012, a total of 712 eggs from 126 wild turtle nests were collected. Eggs were placed in plastic boxes containing moist vermiculite (1:1 ratio with distilled water) and incubated at 26°C (237 eggs), 29°C (237 eggs) or 32°C (238 eggs). Eggs were randomly selected for dissection each week to assess embryos for developmental abnormalities. Embryos were euthanised with an injection of sodium pentobarbital (600 mg/kg).

A replicated study in 2010 in pine forest and artificial conditions in Mississippi, USA (13) found that less than half of artificially incubated gopher tortoise *Gopherus polyphemus* eggs collected from two sites hatched successfully in captivity. Fourteen of 46 (30%) artificially incubated gopher tortoise eggs collected from one site and 22 of 47 (47%) artificially incubated gopher tortoise eggs collected from a second site hatched successfully in captivity. Three hatchlings died within three days of emerging and one never gained function of its rear legs. Ninety-three gopher tortoise eggs were collected from two sites within 24 hours of deposition in May–June 2010 and relocated for artificial incubation. Eggs were placed in dampened vermiculite (0.7 g water to 1.0 g vermiculite) in individual sterilized containers and incubated at 29.3°C. Eggs were collected as part of a head-start programme.

A replicated study in 1994–1999 in New York State, USA (14) found that most wood turtle *Glyptemys insculpta* eggs collected from the wild and artificially incubated hatched successfully. In total, 15 of 18 (83%) artificially incubated wood turtle eggs hatched successfully (1994: 8 of 10 eggs; 1998: 3 of 3 eggs; 1999: 4 of 5 eggs). Wood turtle eggs were collected from wild nests in 1994 (10 eggs), 1998 (3 eggs) and 1999 (5 eggs). The authors noted that eggs were collected from nests that would otherwise have failed. Eggs were placed in dampened vermiculite (1:1 vermiculite: water by weight) and lightly covered with vermiculite in an airtight container, which was opened once a week, until hatching.

A replicated, controlled study in 1974–2012 in a laboratory and 11 wetland sites in New Jersey and Pennsylvania, USA (15) found that bog turtle *Glyptemys muhlenbergii* eggs incubated in a laboratory had higher hatching success than eggs left in wild nests. Hatching success was higher for eggs in the laboratory (74 of 91, 81% [numbers taken from table]) than for eggs in wild nests (caged nests: 42 of 97, 43%; uncaged nests: 53 of 161, 33%). Average hatching date was similar in the laboratory and field (30–31st August). Eggs were transferred from nests to a laboratory and incubated in plastic containers with humus from the wetland.

Incubation temperatures ranged from 26–32°C during the day, and 17–24°C at night. In 1974–1993, a total of 91 eggs from five wetlands were transferred to the laboratory. In 1974–2012, a total of 258 eggs from 11 wetlands were monitored in 27 caged and 55 uncaged nests. Cages were 1 cm wire mesh and buried 8–15 cm into the ground. All eggs were monitored for at least 8–9 weeks to assess hatching success, and hatchlings from the laboratory were released at the original nest site within 5–10 days.

A study in 2015 in desert scrubland in California, USA (16) found that just over half of artificially incubated desert tortoise *Gopherus agassizii* eggs collected from wild adult females hatched in captivity. In total, 74 of 123 desert tortoise eggs hatched after being incubated in captivity (60% emergence success). Eggs were collected from 25 wild adult female desert tortoises in May-June 2015. Eggs were incubated in artificial burrows in an outdoor predator-proof nesting enclosure (in individual 5 x 9 m pens inside a 30 x 30 m enclosure).

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Snakes

- **Four studies** evaluated the effects of relocating nests/eggs for artificial incubation on snake populations. Two studies were in Australia[1,4] and one was in each of Japan[2] and China[3].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (4 STUDIES)

- **Reproductive success (4 studies):** Two studies in Australia[1,4] reported that 87% of carpet python eggs and 83% of brown tree snake eggs hatched successfully following artificial incubation. One study[1] also reported that zero of 10 artificially incubated Oenpelli python eggs hatched. One study in Japan[2] reported that 265 habu eggs[2] hatched successfully following artificial incubation. One replicated, randomized study in China[3] found that hatching success of artificially incubated stripe-tailed ratsnake eggs was lowest at the coolest and warmest temperatures tested.

BEHAVIOUR (0 STUDIES)

OTHER (1 STUDY)

- **Offspring sex ratio (1 study):** One study in Japan[2] reported that artificially incubated habu eggs produced offspring with an even sex ratio.

A study in 1982–1985 in Queensland, Australia (1) found that after bringing brooding female carpet pythons *Morelia spilota* and an Oenpelli python *Morelia oenpelliensis* and/or their egg clutches into captivity and incubating (artificially or with the female) the eggs, some carpet python eggs hatched successfully. From five female carpet pythons that were discovered with clutches of 7–23 eggs (clutch size for one snake not given), 21 of 23 and 5 of 7 eggs hatched successfully (one egg opened artificially), and some eggs from three other females also hatched successfully (number not given). None of the 10 eggs produced by an Oenpelli python hatched successfully. Brooding females that were discovered were brought into captivity along with their clutches, or in one case just the clutch was collected. Some eggs from one clutch were removed surgically. Eggs were incubated either in vermiculite or were left to incubate in the female's coils (see paper for details).

A study in 1981–1992 in the Okinawa Islands, Japan (2) found that artificially incubating habu *Trimeresurus flavoviridis* eggs resulted in some eggs hatching successfully. A total of 265 eggs hatched successfully (total number of eggs not given). When both hatchlings and un-hatched embryos were included, the ratio of females to males was equal (217:234). In 1981–1992, eggs from 62 female snakes were collected. They were incubated at 25–30°C in individual containers containing cotton or sphagnum moss *Sphagnum* sp., saturated with water.

A replicated, randomized study in 1998 in a laboratory in Zhejiang, China (3) found that hatching success of artificially incubated stripe-tailed ratsnake *Elaphe taeniura* eggs was lowest at the coolest and warmest temperatures tested, and that incubation periods and hatchling morphology were also affected by incubation temperature. Hatching success was lower for eggs incubated at the coolest or warmest temperatures (22°C: 6 of 12, 50%; 32°C: 7 of 17, 41%) compared to eggs incubated at intermediate temperatures (24°C: 25 of 32, 78%; 27°C: 19 of 24, 79%; 30°C: 23 of 29, 79%). Incubation period decreased with increased temperatures from 102 days at 22°C to 51 days at 32°C. Five of seven morphological features were also affected by incubation temperature (see paper for more details). In 1998, thirteen captive-born gravid females were acquired and housed in a wire cage (200 x 80 x 80 cm) at 30°C. Eggs were incubated at 22, 24, 27, 30 or 32°C, with eggs from each clutch split evenly between temperatures. Eggs were incubated individually in covered plastic jars in vermiculite and water at a ratio of 1:2. Hatchlings were euthanized by freezing to -15°C to measure a 0range of morphological features.

A study in 1984–1985 at Taronga Zoo, Sydney, Australia (4) found that a wild-caught female brown tree snake *Boiga irregularis* laid eggs in captivity that following artificial incubation hatched successfully. Five of six eggs hatched successfully after an incubation period of 82 days. A gravid female was acquired in 1984 and laid a clutch of eggs soon after arrival. The clutch was incubated artificially but details on incubation conditions are not available.

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Lizards

- **Fifteen studies** evaluated the effects of relocating nests/eggs for artificial incubation on lizard populations. Five studies were in China[6,11-14], two were in each of India[3a,3b], Spain[4,7], the USA[1,10] and New Zealand[5,8] and one was in each of Namibia[2] and Taiwan[9].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (15 STUDIES)

- **Reproductive success (12 studies):** One replicated, controlled study in Namibia[2] found that artificially incubated white-throated savanna monitor eggs had higher hatching success than eggs in natural nests. Three of four studies (including one replicated, controlled study) in the USA[1], Spain[7], Taiwan[9] and China[11] reported hatching success of 56–96% for artificially incubated eggs from wild lizards[1,7,9]. The other study[11] reported that hatching success varied between 11–76%. One replicated, randomized study in India[3b] found that hatching success of artificially incubated garden lizard eggs was lower for eggs incubated in cotton wool compared to those incubated in soil or sand. One replicated study in the USA[10] found that artificially incubated eastern collared lizard eggs that had been laid in captivity in artificial nests had higher hatching success than those laid outside of the artificial nests. Two of five replicated studies (including one randomized, controlled study) in India[3a], Spain[4] and China[6,13,14] found that hatching success of artificially incubated lizard eggs was lower at higher incubation temperatures[3a,4]. Two studies[13,14] found that hatching success was similar across all incubation temperatures. The other study[6] found that hatching success was not affected by temperature fluctuations during artificial incubation.

- **Survival (2 studies):** One replicated, randomized study in New Zealand[5] found that survival of artificially incubated lizards was higher for individuals incubated at higher temperatures. One replicated, randomized study in Spain[4] found that survival of artificially incubated common chameleon hatchlings was affected by incubation temperature but not moisture levels.
- **Condition (7 studies):** Three of five replicated studies (including three randomized studies) in Spain[4], New Zealand[8] and China[6,12,14] found that the size[6,12] or morphology[14] of artificially incubated lizard hatchlings was similar across all incubation temperatures or was not affected by temperature fluctuations[6]. One study[4] found that growth of artificially incubated common chameleon hatchlings was lower for individuals incubated at higher temperatures. The other study[8] found that lizards from eggs incubated at higher temperatures had higher sprint speeds than those incubated at lower temperatures. One replicated, controlled study in Namibia[2] reported that white-throated savanna monitors from artificially incubated eggs were similar in size to hatchlings from natural nests. One replicated, randomized study in India[3b] found that artificially incubating garden lizard eggs in cotton wool, soil or sand resulted in similar sized hatchlings.

BEHAVIOUR (0 STUDIES)

A study in 1979 in the USA (1) found that a wild-caught northern fence lizard *Sceloporus undulatus hyacinthinus* laid eggs that hatched successfully following artificial incubation in captivity. Two weeks after being brought into captivity, a female produced a clutch of 17 eggs. Sixteen of 17 eggs hatched successfully after an incubation period of 45 days. In 1979, a female lizard was captured and housed in a plywood box (3 x 4 x 1 feet) with 11 other northern fence lizards of varying sizes and sexes. The enclosures contained rocks and sand and temperatures ranged from 20–46°C. Eggs were moved to a plastic container with holes drilled in the bottom edge and placed in wet vermiculite. Incubation temperatures were 33–34°C for 20 days, and then varied between 20–29°C for the remainder of the incubation period.

A replicated, controlled study in 1990 in dry savanna in north-central Namibia (2) found that artificially incubated white-throated savanna monitor *Varanus albigularis* egg hatching success was almost double that of natural nests, although hatchlings were similar sized. Overall hatching success of artificially incubated savanna monitor eggs was 81% (120 of 148 eggs) compared to 47% (50 of 107 eggs) for natural nests (results not statistically tested). The average length of artificially incubated hatchlings was similar (115 mm) to hatchlings from natural nests (114 mm; results not statistically tested). Incubation time was longer and hatchlings were larger at lower incubation temperatures and in moister conditions (see original paper for details). In 1990, nine female monitors were radio-tracked through the breeding season. Five females were brought into captivity to lay eggs and four were monitored laying eggs in the wild. Captive-laid eggs were collected and incubated in mixed clutches in 27 boxes containing moistened vermiculite (15–17 eggs/box). Nine boxes contained high (-150 kPa), medium (-550 kPa), or low (-110 kPa) moisture levels. Three boxes of each moisture level were incubated at 27, 29 or 31°C. Artificially incubated (from 148 eggs) and natural-nest hatchlings (from 107 eggs) were checked and weighed on emergence.

A replicated, randomized, controlled study in 1997–1998 in Karnataka, southwest India (3a) found that when eggs from wild-caught garden lizards *Calotes versicolor* were artificially incubated, hatching success was lower at higher incubation temperatures but was not affected by short exposures to high temperatures. Hatching success was lower at 35°C (33 of 59, 53%) and 33°C (36 of 61, 59%) than it was at 30°C (49 of 55, 89%) and at ambient temperatures of 27°C (138 of 148, 93%). Hatching success remained similar when eggs were exposed to short periods at 35°C (1 h/day: 15 of 16, 94%; 3 h/day: 14 of 15, 93%) compared to when eggs were incubated at ambient temperatures of 27°C (15 of 15, 100%). In 1997–1998, gravid female lizards were caught (number not given) and when they reached late gravidity eggs were removed from the oviduct (method not given). Eggs from 15 clutches were split between the treatment temperatures (55 eggs at 30°C, 61 at 33°C and 59 at 35°C) and ambient temperatures (148 eggs at 27°C). A further 46 eggs from three clutches were incubated at 27°C and exposed to 35°C for 1 h/day (16 eggs), 3 h/day (15 eggs) or kept at ambient temperatures (15 eggs). All eggs were incubated in black clay soil, and moisture levels were topped up every two days.

A replicated, randomized study in 1998 in Karnataka, southwest India (3b) found that when eggs from wild-caught garden lizards *Calotes versicolor* were artificially incubated, eggs incubated in cotton wool had lower hatching success than those incubated in soil or sand. Hatching success was lower for eggs incubated in cotton wool (59 of 68, 84%) compared to those incubated in soil (59 of 61, 95%) or sand (84 of 87, 96%). Across all treatments, the average incubation period stayed the same (69 days), and three measures of hatchling size did not differ significantly (body mass, snout-vent length and tail length). In 1997–1998, gravid female lizards were caught (number not given) and when they reached late gravidity eggs were removed from the oviduct (method not given). In 1998, eggs from 15 clutches were incubated in either cotton wool (68 eggs), wet black clay soil (61 eggs) or sand (87 eggs). Eggs were incubated at ambient temperatures of 27°C. Every two days, moisture levels were topped up and new cotton wool was provided.

A replicated, randomized study in 1998–1999 in southern Spain (4) found that during artificial incubation of common chameleon *Chamaeleo chamaeleon* eggs, hatching success, growth and survival were affected by temperature but not by moisture levels. Hatching success was higher at 25°C (100%) than at 29°C (53–76%) but was not affected by moisture levels (wet: 76–100%; dry: 53–100%). Similarly, hatchling survival during the first month was affected by incubation temperature but not moisture levels (result presented as statistical model), and growth was higher for hatchlings incubated at 25°C (0.19 mm/day) compared to those incubated at 29°C (0.09 mm/day). In 1998, ninety-six eggs from six wild nests were split evenly between 16 plastic boxes (6 eggs/box) and completely buried in moist vermiculite. Two temperature treatments (cool: 25°C; warm: 29°C) and two moisture treatments (wet: -150 kPa; dry: -600 kPa) were established, and four boxes each were assigned to each temperature-moisture treatment. All eggs were subjected to an initial low temperature period (<18°C) during November–February before temperatures were gradually increased to the treatment level.

A replicated, randomized study (years not provided) in laboratory conditions in New Zealand (5, same experimental set-up as 8) found that lizard *Oligosoma suteri* hatchlings artificially incubated at 22–26°C survived significantly longer than those incubated at 18°C. Hatchlings incubated at 22°C and 26°C survived for longer (22°C: 94% of 50 survived 18 months; 26°C: 88% of 49 survived 18 months) than those incubated at 18°C (24% of 37 survived 18 months). See original paper for details on the effects of incubation temperature and moisture levels on body size and growth rates. Lizard eggs collected from wild females temporarily brought into captivity were randomly assigned incubation temperatures (18, 22 or 26°C) and moisture levels (–120 and –270 kPa). Eggs were artificially incubated in plastic containers with vermiculite (8–13 eggs/container and 2–4 containers/treatment). After hatching, juveniles were measured and housed in plastic-and-mesh containers for up to 18 months.

A replicated study in 2003 in an artificial setting in Zhejiang, China (6) found that the hatching success of eggs and hatchling size from a species of Chinese skink *Eumeces chinensis* relocated for artificial incubation was not affected by fluctuations in incubation temperatures. At an average temperature of 27°C, hatching success was similar for eggs when incubation temperatures fluctuated by 3°C (12 of 15, 80%) or by 7°C (19 of 22, 86%). A range of hatchling traits, including size, weight and sprint speed were also similar when incubation temperatures fluctuated by 3°C or 7°C (see paper for details). In 2003, four gravid female skinks were brought into captivity and housed in an enclosure (110 x 90 x 50 cm) until they laid their eggs. Eggs were removed and clutches were split evenly between two incubation treatments: average temperature of 27°C with 3°C fluctuations (15 eggs); or average temperature of 27°C with 7°C fluctuations (22 eggs). Temperatures fluctuated gradually over a 24 h period. Hatching success was assessed and hatchling traits were measured (see paper for details).

A replicated, controlled study in 1998 in Cádiz province, southern Spain (7) found that artificially incubating common chameleon *Chamaeleo chamaeleon* eggs resulted in high hatching success, and that hatchling size was affected by the length of an initial cold period during incubation. Overall hatching success was 96%, with just one egg failing during the cold period of incubation (14°C) and two during the warm period (25°C). Hatchling length and body mass were affected by the length of the initial cold period of incubation (results presented as statistical model). In 1998, eggs were collected from seven chameleon nests (82 eggs, 10 or 12 eggs/nest) and incubated in sealed plastic boxes, completely buried in moist vermiculite. Eggs were initially incubated at 14°C for zero, 84, 119 or 149 days (cold period) and then kept at 25°C until hatching (warm period).

A replicated, randomized study (years not provided) in laboratory conditions in New Zealand (8, same experimental set-up as 5) found that incubating lizards *Oligosoma suteri* at higher temperatures resulted in higher sprint speeds at higher ambient temperatures. At ambient temperatures of 26°C, lizards incubated at 22°C (0.8–0.9 m/s) and 26°C (0.9–1.0 m/s) sprinted faster than lizards incubated at 18°C (0.4–0.6 m/s). At temperatures of 18 and 22°C, lizards incubated at 18°C recorded speeds of 0.3–0.4 m/s, lizards incubated at 22°C recorded speeds of 0.6–0.7 m/s, and lizards incubated at 26°C recorded speeds of 0.6–0.8 m/s. The amount of water added to incubation substrate, lizard sex or size did not affect sprint speed (see original paper for details). Lizard eggs from 58 females were collected from the wild and randomly assigned to be incubated at 18 (20 lizards), 22 (49 lizards) and 26°C (48 lizards), at two water potential levels (-120 and -270 kPa). At 4–6 weeks and four months old, lizards were placed on an oval racetrack and encouraged to sprint using a paintbrush at three different ambient temperatures (18, 22 and 26°C). Lizards sprinted three times/ambient temperature. Maximum speeds were recorded and compared.

A study in 2005–2008 in laboratory conditions in Chiayi County, Taiwan (9) found that some wild female Kühne's grass lizards *Takydromus kuehnei* laid eggs in captivity and following artificial incubation, more than half of the artificially incubated eggs hatched successfully. Five females laid a single clutch of 1–2 eggs (9 eggs total; numbers taken from table), and five of nine eggs (56%) hatched successfully. In 2005–2008, a total of 48 lizards were collected from the wild, 19 of which were classified as females. Lizards were housed temporarily (< 5 days) in small glass cages (25 x 25 x 27 cm) with a substrate of clean river sand overlain with sphagnum moss *Sphagnum* sp. Eggs were removed and placed in a small-animal cage on a substrate of clean river sand covered with sphagnum moss. Eggs were incubated at room temperature (25–32°C) and cages were misted to maintain a moist environment. Adult lizards were re-released in the same area they had been captured.

A replicated study in 2011 in laboratory conditions in Oklahoma, USA (10) found that some artificially incubated eggs from wild-caught, gravid female eastern collared lizards *Crotaphytus collaris* that laid eggs in captivity hatched successfully, but only eggs that were laid inside artificial nests hatched. All 17 wild-caught gravid female eastern collared lizards laid eggs in captivity (one clutch/individual, 5–9 eggs/clutch). Twelve lizards laid eggs inside artificial nest chambers (74 total eggs) and these eggs had a 62% hatching success after artificial incubation (46 of 74 eggs hatched). Five lizards laid eggs outside of artificial nest chambers (29 total eggs) and none of these eggs hatched after artificial incubation (23 eggs were desiccated when found after being laid and six eggs became mouldy during incubation). Seventeen gravid female lizards were caught in the Glass Mountains and moved to a laboratory where they were housed individually in partitioned wooden and metal-mesh cages. Each cage section (80 x 40 x 40 cm) contained gravel substrate, artificial lighting and an artificial nest made from bricks and sand/peat moss (see original paper for details). Lizards were fed and watered regularly. Eggs were moved for artificial incubation within 16 hours of being laid and adult lizards were returned to their capture site.

A replicated study in 2011 in Gansu, China (11) found that after bringing females of two species of toad-headed agamas *Phrynocephalus przewalskii* and *Phrynocephalus versicolor* into captivity to lay eggs, less than half artificial incubated eggs hatched successfully, and hatching success was lower at the highest incubation temperature. Hatching success was lowest at the highest incubation temperature for both species (*Phrynocephalus przewalskii*: 34°C: 32–36%; 26–30°C: 40–53%; *Phrynocephalus versicolor*: 34°C: 11–22%; 26–30°C: 52–76%), although this result was not statistically tested. Moisture content of the incubation medium had no effect on hatching success. Incubation period decreased at higher temperatures for both species (26°C: 44–46 days; 30°C: 31–33 days; 34°C: 23–26 days). In 2011, wild female lizards of both species were captured and housed in groups of 15 in cages (800 x 360 x 400 mm) with a sand substrate. Temperatures of 25–37°C were available during the day and were 20°C at night. Eggs were collected (*Phrynocephalus przewalskii*: 263 eggs from 101 females; *Phrynocephalus versicolor*: 185 eggs from 66 females) and assigned to three temperature (26, 30, 34°C) and two moisture level (2 g water/5 g vermiculite, 2 g water/8 g vermiculite) treatments. Eggs were incubated in plastic containers (150 ml).

A replicated study in 2010–2011 in Gansu and Zhejiang provinces, China (12) found that after bringing wild, gravid females of five *Phrynocephalus* species into captivity, some artificially incubated eggs hatched successfully. Data on hatching success is not provided. Incubation period was shorter at higher temperatures for all species (24°C: 44–56 days; 28°C: 32–38 days: 32°C: 26–28 days), and none of seven measures of hatchling size were affected by temperature (results presented as statistical model). In 2010–2011, gravid females of five species were collected and housed in cages (900 x 650 x 600 mm) in groups of 7–10 individuals of the same species. A substrate of sand, and clay tiles were provided, and temperatures were 20–28°C. Females were moved to individual cages (200 x 200 x 200 mm) to lay eggs. Eggs were incubated at 24°C (54 eggs), 28°C (44 eggs) or 32°C (59 eggs) in individual covered jars (50 ml) in vermiculite (1:1 with water by weight). Adult females were released back in to the wild after 3–4 months.

A replicated, randomized study in 2010–2011 in laboratory conditions in Zhejiang province, China (13) found that most eggs laid in captivity by wild Chinese skinks *Plestiodon chinensis* and artificially incubated hatched successfully, and hatching success was not affected by incubation temperature. Overall hatching success was 86% (837 of 972 eggs) and was similar at all incubation temperatures (83–86%). In addition, incubation period decreased at higher temperatures (40 days at 24°C vs 19 days at 32°C). In 2010–2011, seventy-two gravid females were brought into captivity and housed in groups of 5–8 in enclosures (1.5 x 1.5 x 0.6 m) with a turf-covered substrate. A total of 972 viable eggs were collected and placed in individual plastic jars (50 ml) with moist vermiculite (-12 kPa). Clutches of eggs were divided between five constant temperature treatments (24, 26, 28, 30 or 32°C) or one fluctuating treatment (incubated outside). After laying, adult females were re-released at their point of capture.

A replicated, randomized study in 2013–2015 in south China (14) found that artificially incubating forest skink *Sphenomorphus incognitus* eggs at different temperatures did not affect hatching success or hatchling morphology, but that higher temperatures resulted in shorter incubation periods. Hatching success did not change significantly at different incubation temperatures (69% at 22°C; 77% at 25°C; 82 % at 28°C) or when incubation temperature fluctuated around an average of 25°C (3°C fluctuation: 79%; 5°C fluctuation: 64%). Five measures of morphology were also similar at different incubation temperatures (see paper for details). Average incubation period varied between 76 days at 22°C and 40 days at 28°C. In 2013–2015, twenty-seven wild, gravid female skinks were collected and housed in individual plastic cages (540 x 400 x 320 mm) with a substrate of moist soil at 20–28°C. A total of 136 eggs were incubated in individual, covered plastic jars (50 ml) with moist vermiculite. Eggs from each clutch were divided equally between three constant temperature treatments (13 eggs at 22°C, 26 at 25°C, 11 at 28°C) and two treatments that fluctuated around an average of 25°C (28 eggs at 3°C fluctuation, 25 at 5°C fluctuation).

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Crocodilians

- **Six studies** evaluated the effects of relocating nests/eggs for artificial incubation on crocodilian populations. Two studies were in the USA[3,5] and one study was in each of Zimbabwe[1], Argentina[2], Venezuela[4] and Australia[6].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (6 STUDIES)

- **Reproductive success (5 studies):** Two replicated studies in Zimbabwe[1] and the USA[5] reported that hatching success for 20,000 Nile crocodile eggs[1] and >30,000 American alligator eggs[5] that were artificially incubated was 74%[1] and 61%[5]. Two studies (including one replicated study) in Argentina[2] and Venezuela[4] reported that 43–100% of road-snouted caiman eggs[2], 66% of American crocodile eggs and 54% of Orinoco crocodile eggs[4] hatched successfully following artificial incubation. One replicated, before-and-after study in Australia[6] reported that hatching success of artificially incubated saltwater crocodile eggs differed when the project was under local compared to external management.
- **Condition (1 study):** One replicated, controlled, paired study in the USA[3] found that American alligator eggs relocated for artificial incubation produced larger hatchlings than eggs left in situ.

BEHAVIOUR (0 STUDIES)

A replicated study in 1967–1974 in three rearing stations along the Zambezi River in Zimbabwe (1) found that three-quarters of artificially incubated Nile crocodile *Crocodylus niloticus* eggs hatched in captivity. Over seven years, artificially incubated Nile crocodile egg hatching success was 74% (16,697 of 22,697 eggs hatched). The authors reported that collecting eggs very soon after laying had a detrimental effect on hatching success. Nile crocodile eggs were collected from the wild, hatched and reared in three rearing stations (at Kariba Lake, Binga and Victoria Falls) as part of a crocodile farming initiative in 1967–1973 (128–2,475 eggs collected/station/year). Eggs were artificially incubated in captivity (no details are provided).

A replicated study in 2001–2002 in a laboratory in Santa Fe province, Argentina (2) found that artificially incubated broad-snouted caiman *Caiman latirostris* eggs hatched in captivity. Hatching success of artificially incubated broad-snouted caiman eggs taken from the wild ranged from 43–100% (hatching success of seven caiman nests: 30 of 36 eggs hatched; 18 of 37 eggs hatched; 35 of 41 eggs hatched; 20 of 30 eggs hatched; 13 of 30 eggs hatched; 18 of 26 eggs hatched; 35 of 35 eggs hatched). Between 1990 and 2002, a head-starting programme collected caiman eggs from wild nests (December–January), artificially incubated the eggs and reared hatchlings for up to nine months before releasing caiman (with individual scale markings) back into the collection site (see original paper for details). In austral summer 2001–2002, clutches from seven head-started female broad-snouted caiman were collected from the wild (26–41 eggs/nest) and artificially incubated (at 31.5°C and 95% relative humidity) until hatching.

A replicated, controlled, paired study in 1999–2004 in an area of marsh in Louisiana, USA (3) found that relocating American alligator *Alligator mississippiensis* eggs for artificial incubation soon after laying resulted in heavier, longer hatchlings compared to eggs left in the nests until just before hatching. Eggs relocated soon after laying produced heavier (after hatching: 39–56 g; 6–9 months old: 795–1,270 g) and longer (after hatching: 24–26 cm; 6–9 months old: 63–78 cm) hatchlings than eggs from naturally incubated nests (mass: after hatching: 37–53 g; 6–9 months old: 795–1,130 g; length: after hatching: 23–26 cm; 6–9 months old: 62–74 cm). Alligator nests were located by helicopter and ground surveys (three nests in 1999 and six nests in 2003). Half of each clutch was relocated for artificial incubation soon after laying, whereas the other half was left in the nest and collected just prior to hatching. Relocated eggs were artificially incubated at 31–32°C. Weight and length of all hatchlings was measured two days after hatching, and then three more times at 2–3 month intervals.

A study in 2009 in two river basins in Venezuela (4) found that most American crocodile *Crocodylus acutus* and Orinoco crocodile *Crocodylus intermedius* eggs hatched successfully after being collected from the wild and artificially incubated. Results were not statistically tested. In total 66% of American crocodile eggs (200 of 305 eggs) and 54% of Orinoco crocodile eggs (116 of 216 eggs) hatched successfully after artificial incubation. Egg collection was carried out in 2009 (521 eggs collected overall) in the Santa Ana (305 American crocodile eggs) and Manapire (216 Orinoco crocodile eggs) river basins. Eggs were transported to a brick-walled building in each location (25 m^2 and 5 m^2 in size) with a zinc roof closed to predator access. The larger incubation room temperature was maintained at 32°C by five light bulbs. Eggs were stored in sand-filled insulated polystyrene boxes. The sand was kept damp by adding water at regular intervals.

A replicated study in 2007–2012 in hatching facilities across six counties in Texas, USA (5) found that artificially incubating American alligator *Alligator mississippiensis* eggs resulted in more than half of eggs hatching successfully. Average hatching success was 61% (average of 23 of 37 eggs/nest) and hatching success of viable eggs was 71% (average of 23 of 32 viable eggs/nest). In 2007–2012, a total of 33,454 eggs were collected from 902 wild alligator nests, and the viability of eggs was determined by examining egg colour, odour and presence of an opaque band. Eggs and nesting materials were transported in wire baskets to hatching facilities, where they were incubated at 31–32.8°C and 100% humidity, buried inside the nesting material. Eggs were removed from 50% of nests that were discovered during surveys, and surveys were carried out on foot, by boat and by helicopter.

A replicated, before-and-after study in 1989–2015 in hatching facilities within four river systems in Northern Territory, Australia (6) found that artificially incubated saltwater crocodile *Crocodylus porosus* eggs hatched in captivity, but hatching success rates differed between local and external management. Results were not statistically tested. Hatching success of saltwater crocodile eggs as part of a sustainable harvest programme was 49% when run by a local Indigenous community organisation (654 hatchlings from 1,396 live eggs/year) compared to 84% when it was run by an external management company (1,413 hatchlings from 1,659 live eggs/year). Saltwater crocodile eggs were collected and incubated as part of a regional government-led sustainable harvest initiative. In 1989–1997 an external management company ran the programme. In 1998–2015 it was run by a local Indigenous management company. In 1996–1997 eggs were harvested by the external company and incubated by the Indigenous management company. There was no harvest in 2007–2008. Annual quotas were 2,700–3,000 eggs/year (total limit of 70,000 eggs/year across the territory). Eggs were incubated at a constant temperature of 32°C and ≥99% humidity. Local workers were paid based on the number of eggs collected and hatchlings produced.

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Tuatara

- **Two studies** evaluated the effects of relocating nests/eggs for artificial incubation on tuatara populations. Both studies were in New Zealand[1,2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Reproductive success (2 studies):** One of two replicated studies (including one controlled study) in New Zealand[1,2] reported that hatching success of tuatara eggs relocated for artificial incubation was 86–100%[1]. The other study reported hatching success of 44%[2].
- **Condition (1 study):** One replicated, controlled study in New Zealand[1] found that 10 months after hatching, artificially incubated tuatara were larger that those from natural nests.

BEHAVIOUR (0 STUDIES)

A replicated, controlled study in 1998 in a captive setting and an island in Marlborough Sounds, New Zealand (1) found that tuatara *Sphenodon punctatus* eggs relocated for artificial incubation had high hatching success and that 10 months after hatching, young were larger than those from naturally incubated nests. Hatching success for artificially incubated eggs was 86–100% (18°C: 105 of 120, 86%; 21°C: 80 of 80, 100%; 22°C: 113 of 120, 94%) and all but three hatchlings survived for at least 10 months. Just after hatching, artificially incubated tuatara were larger in two of five measures and similar in three of five measures compared to tuatara that were naturally incubated for 11 months, but 10 months after hatching, artificially incubated tuatara were larger in all five measures (see paper for details). In 1998, a total of 320 eggs were collected either from natural nests (154 eggs from 29 clutches) or by inducing females to lay eggs with oxytocin (166 eggs from 21 clutches). Eggs were incubated in moist vermiculite in plastic containers, with clutches divided equally for incubation at 18°C, 21°C or 22°C. In addition, eggs from 25 naturally laid nests were left in situ for 11 months and then eggs and hatchlings were brought into captivity (eggs were incubated at 22°C until hatching). Hatching success was monitored and all hatchlings were weighed and measured.

A replicated study in 1990–2007 in artificial enclosures in North Island, New Zealand (2) found that less than half of relocated artificially incubated wild tuatara *Sphenodon punctatus* eggs hatched. Over 16 years, 44% of eggs (241 of 553 eggs) laid by wild tuatara in captivity and relocated for artificial incubation hatched successfully. The first clutches to hatch successfully were laid 2–8 years after tuatara were brought into captivity. Second-generation female hatchlings that had been artificially incubated went on to produce three clutches during the study. In 1990–1992 four entire tuatara populations from four islands (6–15 individuals/island) were placed in one of three captive facilities pending eradication of pacific rats *Rattus exulans*. Clutches laid by 15 females were moved to a separate facility for artificial incubation in dampened vermiculite at temperatures to ensure an even sex ratio (see original paper for details). Four clutches were induced and the remaining 27 were laid naturally. Eggs that perished shortly after being laid (5–16 eggs in 2 clutches) and eggs laid by artificially incubated females were excluded from the data.

(1) Nelson N.J., Thompson M.B., Pledger S., Keall S.N. & Daugherty C.H. (2004) Egg mass determines hatchling size, and incubation temperature influences post-hatching growth, of tuatara *Sphenodon punctatus*. *Journal of Zoology*, 263, 77–87.

(2) Keall S.N., Nelson N.J. & Daugherty C.H. (2010) Securing the future of threatened tuatara populations with artificial incubation. *Herpetological Conservation and Biology*, 5, 555–562.

14.24. Recover eggs from injured or dead reptiles

- **Two studies** evaluated the effects of recovering eggs from injured or dead reptiles on their populations. One study was in each of the USA[1] and Columbia[2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Reproductive success (2 studies):** One replicated, controlled study in Columbia[2] found that eggs recovered from harvested Magdalena river turtles had similar hatching success compared to both relocated and natural turtle nests. One replicated study in the USA[1] found that 64% of eggs recovered from road-killed red-eared sliders hatched successfully.

BEHAVIOUR (0 STUDIES)

Background

When gravid female reptiles are killed, for example as a result of harvesting or as road collisions, it may be possible to collect eggs from carcasses for incubation in captivity.

Studies that discuss artificially incubating reptile eggs collected from wild-laid nests are included under *Relocate nests/eggs for artificial incubation*.

A replicated study in a laboratory in Illinois, USA (1) found that after incubating eggs recovered from road-killed red-eared sliders *Trachemys scripta elegans*, more than half of the eggs hatched successfully, and hatching success was higher for eggs from turtles found with intact shells compared to those with open shells. Forty-three of 67 (64%) eggs hatched successfully, and hatching success was higher for eggs recovered from turtles with intact shells (30 of 35, 86%) compared to those with open shells (13 of 32, 41 %). Of 32 turtles that were found on a road having been hit by a vehicle, nine contained 2–21 unbroken eggs. One turtle survived and was later released after laying eggs. Unbroken eggs were transferred to a laboratory and partially buried in perlite incubation medium in plastic containers (32 x 19 x 10 cm), with aluminium foil layered under the lid. Clutches were incubated separately. A road was searched for turtles hit by vehicles at least twice daily during the nesting season (months not given).

A replicated, controlled study in 2005–2006 in one wetland and two riverbank sites in northern Columbia (2) found that hatching success of Magdalena river turtles *Podocnemis lewyana* was similar for eggs recovered from harvested adult turtles compared to eggs from relocated nests and natural nests. Hatching success was statistically similar for eggs recovered from harvested turtles and buried in artificial nests (21%) compared to those from relocated nests (58%) and natural nests (41%). In 2005, seven clutches of eggs were recovered from turtles that had been harvested by local people and incubated in artificial nests that were dug into the riverbank. In 2005–2006, a further 24 nests were relocated higher up the beach away from rising river levels and 22 nests were left in place. All nests were covered with wire mesh cylinders (1 x 1 cm) that were 40 cm wide and 50 cm high, with a 3 x 3 cm plastic mesh on top. In February–May 2005–2006, beaches were searched daily, with the aid of dogs *Canis lupus familiaris*, to locate turtle nests. All nests were inspected daily and excavated after hatching, or after 74 days of incubation.

(1) Tucker J.K. (1995) Salvage of eggs from road-killed red-eared sliders, *Trachemys scripta elegans*. *Chelonian Conservation and Biology*, 1, 317–318.
(2) Correa-H J.C., Cano-Castaño A.M., Páez V.P. & Restrepo A. (2010) Reproductive ecology of the Magdalena River turtle (*Podocnemis lewyana*) in the Mompos Depression, Colombia. *Chelonian Conservation and Biology*, 9, 70–78.

Protection of breeding adults

14.25. Bring threatened wild populations into captivity

- **Three studies** evaluated the effects on reptile populations of bringing threatened wild populations into captivity. One study was in each of New Zealand[1], Myanmar[2] and Australia[3].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (3 STUDIES)

- **Abundance (2 studies):** One of two replicated studies in Myanmar[2] and Australia[3] found that after bringing Burmese start tortoises[2] into captivity the populations increased from 175 individuals to over 7,000 in 12 years. The other study[3] found that Lister's gecko and blue-tailed skink populations remained stable or grew over 4–5 years in captivity.

- **Reproductive success (2 studies):** Two replicated studies in New Zealand[1] and Myanmar[2] found that after bringing tuatara[1] and Burmese start tortoises[2] into captivity, 44% of tuatara eggs hatched successfully in 16 years, and the number of hatchlings produced by Burmese start tortoises increased from 168 to over 2,000 in eight years[2].
- **Survival (1 studies):** One replicated study in New Zealand[1] found that varying proportions of wild tuatara brought into captivity survived for 16 years.

BEHAVIOUR (0 STUDIES)

Background
Sometimes it may be necessary to bring all reptiles from a single population into captivity, temporarily or permanently, as a last resort to prevent a species from becoming extinct in the wild. These captive populations are often called 'captive assurance colonies' (Buhlmann *et al*. 2012). Consideration must be given to maintaining captive assurance colonies as genetically-viable breeding groups in order to maximise the chance of the species surviving.

Buhlmann K.A., Hudson R., & Rhodin A.G. (2002) *A global action plan for conservation of tortoises and freshwater turtles: strategy and funding prospectus 2002–2007*. Conservation International and Chelonian Research Foundation, Washington D.C.

A replicated study in 1990–2007 in three captive facilities on North Island, New Zealand (1) found that most wild tuatara *Sphenodon punctatus* brought into captivity survived and bred. Over 16 years, eight of eight and six of six tuatara from two island populations survived in captivity. In addition, 11 of 15 tuatara from a third island survived and were released back into the wild (the fate of the remaining four is not described) and five of 11 tuatara from a fourth island survived in captivity. Clutches were laid in 13 of 16 years by 15 of 22 females, 44% of eggs hatched (241 of 553 eggs) and second-generation females produced three clutches. In 1990–1992, entire populations of tuatara from four islands were captured (6–15 individuals/island) and placed in one of three captive facilities pending eradication of pacific rats *Rattus exulans*. Tuatara were housed in predator-proof outdoor enclosures. In 1992–2007, eggs were moved to a separate facility and artificially incubated (see original paper for details). Hatchlings were returned to their source facility after one week to11 months. Hatching success does not include eggs that perished shortly after being laid (5–16 eggs in 2 clutches) and eggs laid by second-generation females.

A replicated study in 2004–2016 in three wildlife sanctuaries in the central dry zone of Myanmar (2) found that three captive assurance colonies of Burmese star tortoises *Geochelone platynota* survived at least 12 years in captivity and bred. The total population of three captive assurance colonies of Burmese start tortoises increased from approximately 175 tortoises in 2004 to 7,150 tortoises in October 2016 (≤ 2-years-old: 4,849 individuals; subadults: 1,794 individuals; breeding adults: 501 individuals). Over 12 years, hatching rates were 50–75% (no further details are provided) and total annual number of hatchlings produced increased from 168 individuals in 2008 to 2,142 individuals in 2016. Female hatchlings hatched before 2010 had started laying eggs by 2016. The Burmese star tortoise was considered ecologically and functionally extinct in the wild during the 2000s. In 2004, three wildlife sanctuaries located within the tortoise's historical geographic range were established as captive assurance colonies using confiscated juvenile, subadult and adult tortoises and some wild tortoises as founding stock (approximately 175 total tortoises of an equal sex ratio). Tortoises were housed in electric-fenced outdoor enclosures with shelter, food and water provided (see original paper for husbandry details). Nesting activity was monitored and eggs incubated and hatched in situ.

A replicated study in 2009–2016 in two captive-breeding programmes on Christmas Island and at Taronga Zoo, Australia (3) found that after bringing wild Lister's geckos *Lepidodactylus listeri* and Christmas Island blue-tailed skinks *Cryptoblepharus egeriae* into captivity, populations were maintained successfully in four of four cases. Results were not statistically tested. On Christmas Island, populations of Lister's gecko grew from 50 in 2012 to 500 in 2016, and populations of blue-tailed skinks grew from 150 in 2012 to 750 in 2016. At Taronga zoo, populations of Lister's gecko (70 in 2011 and 70 in 2016) and blue-tailed skinks (100 in 2011 and 220 in 2016) remained relatively stable. In 2009, all Lister's geckos and blue-tailed skinks that could be found on Christmas Island were brought into captivity. From these wild-caught individuals and their offspring, 56 geckos and 83 skinks were transported to Taronga, and the remaining 70 geckos and 109 skinks were maintained at facilities on Christmas Island. Captive management aimed to maximise retention of genetic diversity (see paper for more details).

(1) Keall S.N., Nelson N.J. & Daugherty C.H. (2010) Securing the future of threatened tuatara populations with artificial incubation. *Herpetological Conservation and Biology*, 5, 555–562.

(2) Platt S.G., Platt K., Khaing L.L., Yu T.T., Aung S.H., New S.S., Soe M.M., Myo K.M., Lwin T., Ko W.K., Aung S.H.N. & Rainwater T.R. (2017) Back from the brink: Ex situ conservation and recovery of the critically endangered Burmese star tortoise (*Geochelone platynota*) in Myanmar. *Herpetological Review*, 48, 570–574.

(3) Andrew P., Cogger H., Driscoll D., Flakus S., Harlow P., Maple D., Misso M., Pink C., Retallick K., Rose K., Tiernan B., West J. & Tiernan B. (2018). Somewhat saved: a captive breeding programme for two endemic Christmas Island lizard species, now extinct in the wild. *Oryx*, 52, 171–174.

14.26. Fence cliff edges to prevent individuals from falling

- **One study** evaluated the effects on reptile populations of fencing cliff edges to prevent individuals from falling. This study was in Australia[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Survival (1 study):** One before-and-after study in Australia[1] found that after installing a fence along a small cliff edge, fewer green turtle carcasses were found at the base of the cliff compared to before installation.

BEHAVIOUR (0 STUDIES)

Background

Installing fencing along cliff edges in areas frequently used by reptiles may reduce the risk of individuals falling from cliffs. Fencing may have unintended consequences on other species, and so the potential costs and benefits of fencing should be carefully considered.

A before-and-after study in 2010–2012 in a vegetated coral cay in Raine Island, the Great Barrier Reef, Australia (1) found that fencing cliff edges to prevent green turtle *Chelonia mydas* falls reduced mortality. Fewer turtle carcasses were found at the base of the cliff after the fence was installed (2 carcasses) compared to before installation (60 carcasses). Fencing totalled 100 m of modified aluminium pool fencing (50 cm high with vertical bars at 30 cm intervals), which was installed in three areas around the tops of small cliffs on the eastern end of the island in November 2011. The fencing blocked off the cliff edges but kept open natural ramps for entrance and egress of turtles to the centre of the island. Surveys of the number and location of sea turtle carcasses on the island occurred in November 2011 (before installation) and December 2011 and February 2012 (after installation).

(1) Great Barrier Reef Marine Park Authority (2012) *Raine Island Adaptive management to conserve marine turtles. Adaptation plans for islands, Final Bulletin, 24/05/2012*. Climate Change Group, Great Barrier Reef Marine Park Authority.

14.27. Provide rewards (monetary or non-monetary) for reporting injured or entangled reptiles

- We found no studies that evaluated the effects of providing rewards for reporting injured or entangled reptiles on their populations.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background
A reward system may increase the incentive to report injured or entangled reptiles, and as such increase the number of these reptiles that may be assisted/rescued. Before implementing any rewards system, the potential for unintended consequences should be carefully considered.

14.28. Provide reptiles with escape routes from canals, drains and ditches

- We found no studies that evaluated the effects on their populations of providing reptiles with escape routes from canals drains and ditches.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background
Reptiles may be attracted to canals, drains and ditches for drinking, or may need to cross such obstacles while dispersing through the landscape. When such waterways have steep sides, reptiles may fall in and be unable to escape, or aquatic reptiles may enter deliberately but struggle to exit the water. Escape routes may be installed to enable reptiles that have fallen in or otherwise entered the water

to escape back onto land. These may take the form or ramps, ladders, shallow inlets or other structures that reptiles could use to climb out.

Supplementary feeding in the wild

14.29. Provide supplementary food or water

- **Four studies** evaluated the effects of providing supplementary food or water on reptile populations. Two studies were in the USA[2,3] and one was in each of Indonesia[1] and Australia[4].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Survival (2 studies):** One replicated, controlled study in the USA[3] found that translocated desert tortoises given supplementary water had similar survival over two years compared to those given no supplementary water.
- **Reproduction (1 study):** One randomized, controlled study in the USA[2] found that more Western diamond-backed rattlesnakes provided with supplementary food reproduced compared snakes that were not fed.
- **Condition (2 studies):** Two controlled studies (including one randomized and one replicated study) in the USA[2,3] found that Western diamond-backed rattlesnakes[2] or translocated desert tortoises[3] that were given supplementary food[2] or water[3] grew more than those that were not supplemented.

BEHAVIOUR (4 STUDIES)

- **Use (1 studies):** One controlled, before-and-after study in Indonesia[1] found that areas where supplementary food was provided were used more frequently by Komodo dragons than other parts of the island.
- **Behaviour change (3 studies):** One of two controlled studies (including one replicated, before-and-after study) in the USA[2] and Australia[4] found that that Pygmy bluetongue lizards translocated into enclosures and given supplementary food showed differences in three behaviour measures compared to lizards given no food[4]. The other study[2] found that fed and unfed Western diamond-backed rattlesnakes showed similar behaviours across four measures. One replicated, controlled study in the USA[3] found that translocated desert tortoises given supplementary water moved longer distances than those given no supplementary water.

Background

Providing supplementary food could have a positive effect on wild reptile populations, particularly where natural food sources in the wider environment are scarce. Supplementary food may be used to support small, vulnerable populations, including in the context of translocations and reintroductions. Translocated or released reptiles may be especially vulnerable immediately after release, while they struggle to find natural food in an unfamiliar area. Furthermore, if the time they spend looking for food is increased, this may make them more vulnerable to predation. Hence, providing supplementary food and/or water at and after the period of release may improve longer term survival.

Supplementary food may also be provided in the context of wildlife tourism operations. However, under these circumstances care should be taken to ensure that the provision of potentially unnecessary food does not lead to unintended negative consequences for wild populations.

A controlled, before-and-after study in 1990–1996 on two tropical islands in Komodo National Park, Indonesia (1) found that providing supplementary food at one location for Komodo dragons *Varanus komodoensis* resulted in larger numbers at the feeding site, but no evidence of an increase in the total population size. Results were not statistically tested. During four years of supplementary feeding (1990–1993), average daily dragon numbers at the feeding site were 16–19 dragons/day. After feeding ceased, average numbers were 13 in 1994 and six in 1995, by which point they were similar to numbers recorded at baited survey locations across the islands (3–4 dragons/day) in 1993–1995. Goat carcasses were provided two times/week from 1990 to August 1994 at a tourist viewing platform. In addition, the population was censused over 24 hours annually in October by securing a dead goat to permanent plots at 47 locations on Komodo island and 29 locations on Rinca island.

A randomized, controlled study in 2002–2003 in one desert site in Arizona, USA (2) found that providing supplementary food for Western diamond-backed rattlesnakes *Crotalus atrox* resulted in more snakes giving birth and faster growth compared to unfed snakes. More fed snakes reproduced over the 19-month period (7 of 9 snakes; 5 young/litter) than did unfed snakes (1 of 8 snakes; produced 2 young). Fed snakes grew faster than unfed snakes (fed: 0.4 cm/month and 4 cm total growth; unfed: 0.1 cm/month and 1 cm total growth) and gained more mass (fed: 21 g/month; unfed 1 g/month). In general, body condition of fed and unfed snakes was similar, though after giving birth, fed snakes had better body condition (see paper for details). Four measures of above ground activity and home range size were similar for fed and unfed snakes (see paper for details). In March 2002, seventeen wild female snakes were implanted with radio transmitters and released back into the wild. Nine were selected to received supplementary feeding, and eight received no additional food. Fed snakes were offered thawed rodents 1–4 times/week. Snakes were located 1–5 times/week during the active seasons (March–November) and 1–2 times/month during winter (November–March).

A replicated, controlled study in 1997–1998 in a site of desert scrub in southern Nevada, USA (3) found that translocated desert tortoises *Gopherus agassizii* provided with supplementary water had similar survival but moved more and grew more than non-supplemented tortoises. Mortality rates were similar between supplemented (4 of 15, 27%) and non-supplemented (2 of 13, 15%) translocated tortoises in the year of release. No tortoises died in the second year. Water supplemented tortoises grew more (0.0014 mm/day) and moved longer distances (up to 3,800 m, males only) compared to non-supplemented tortoises (0.007 mm growth/day and 700 m, males only). Released tortoises were held in outdoor pens for two (juveniles) to seven (adults) years, having been removed from areas undergoing urban development. One to two months prior to release, tortoises either received supplementary water (sprinklers on for 15 minutes/day and saucers placed to catch water; 6 females, 8 males, 1 juvenile) or received no water (7 females, 5 males, 1 juvenile). Tortoises were released into artificial burrows in April–May 1997, and the release site was fenced off from a nearby road. Tortoises were relocated by radio-tracking through July 1997 to November 1998.

A replicated, controlled, before-and-after study in 2009 in grass, bare ground and tilled soil enclosures in southern Australia (4) found that translocated pygmy bluetongue lizards *Tiliqua adelaidensis* provided with supplementary food and artificial burrows were less active, spent less time basking outside of burrows and were observed less in bare ground habitat than unfed lizards. Fed lizards were active for 1.5 hours less each day (4 hours) than unfed lizards (5.5 hrs). Fed lizards spent less time basking at their burrow entrance (19 minutes/hour) compared to unfed lizards (29 minutes/hour). Fed lizards were observed less frequently in bare ground habitat on most days and this effect became larger towards the end of the feeding period (see original paper for details). In total 16 lizards were captured and moved to a trial site in a zoo. Four lizards were released into four 15 m enclosed cages in November 2009. Cages contained short grass, bare ground and tilled soil. Lizards were fed mealworms daily in burrows for seven days in two cages and not fed in the other two cages, then no lizards were fed for two days before the feeding regime began again, but this time the previously unfed cages were fed daily for seven days and the other cages were not. Artificial burrows were built from hollowed wooden poles (30 cm long, 3 cm diameter) pushed into grassy or tilled soil (82 burrows/cage). Lizards were monitored by four surveillance cameras/cage during daylight hours from the second to seventh days of the feeding regime (12 days total).

(1) Walpole M.J. (2001) Feeding dragons in Komodo National Park: a tourism tool with conservation complications, *Animal Conservation,* 4, 67–73.

(2) Taylor E.N., Malawy M.A., Browning D.M., Lemar S.V. & DeNardo D.F. (2005) Effects of food supplementation on the physiological ecology of female western diamond-backed rattlesnakes (*Crotalus atrox*). *Oecologia*, 144, 206–213.

(3) Field K.J., Tracy C.R., Medica P.A., Marlow R.W. & Corn P.S. (2007) Return to the wild: translocation as a tool in conservation of the desert tortoise (*Gopherus agassizii*). *Biological Conservation*, 136, 232–245.

(4) Ebrahimi M. & Bull C.M. (2012) Food supplementation reduces post-release dispersal during simulated translocation of the endangered pygmy bluetongue lizard *Tiliqua adelaidensis*. *Endangered Species Research*, 18, 169–178.

15. Education and awareness raising

Background

This chapter includes interventions focussed on education and awareness campaigns in response to a range of threats, as well as studies that measure the effect of an intervention carried out to change human behaviour for the benefit of reptile populations. Studies do not always measure the effect of these interventions on reptile populations, and therefore we also include those that measure the impact on actual or intentional human behaviour.

It should be noted that there are many complex factors that influence human behaviour and providing education does not guarantee that behaviour will change. It may be necessary to collaborate with social scientists to design appropriate education programmes that consider the attitudes, values and social norms of the target audience.

Studies describing educational campaigns in response to specific threats are described in the chapter on that threat.

15.1. Use education and/or awareness campaigns to improve behaviour towards reptiles and reduce threats

- **Seven studies** evaluated the effects of using education and/or awareness campaigns to improve behaviour towards reptiles and reduce threats. One study was in each of Costa Rica[1], India[2], the Philippines[3], Dominica[4], the USA[5], Saint Kitts[6] and Colombia[7].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (3 STUDIES)

- **Abundance (1 studies):** One before-and-after study in the Philippines[3] found that following a communication, education, and public awareness campaign, the population of Philippine crocodiles increased.
- **Reproductive success (1 study):** One study in Costa Rica[1] found that during a community-based education programme the percentage of leatherback turtle nests lost to poaching decreased.
- **Survival (3 studies):** Two before-and-after studies in the Philippines[3] and Dominica[4] found that following education and awareness campaigns, one in combination with use of road signs[4], human killing of Philippine crocodiles[3] decreased and there were fewer road-deaths of lesser Antillean iguanas[4] compared to before the campaigns began. One study in India[2] reported that following education and awareness campaigns in combination with creating a network of local snake experts[2], local snake experts reported that they intervened to save 276 non-venomous snakes from being killed over six years.

BEHAVIOUR (0 STUDIES)

OTHER (5 STUDIES)

- **Human behaviour change (3 studies):** One replicated study in Colombia[7] found that in areas with conservation initiatives relating to turtles, more people reported changing consumption habitats and fewer people reported using turtles for food compared to in areas with no initiatives, however, stated rates of hunting, buying and selling of turtles remained similar One study in Saint Kitts[6] found that attending an educational summer camp on turtle conservation had mixed effects on reported behaviours in relation to sea turtles of attendees and their parents/guardians, and mixed effects on whether they took part in conservation activities after the camp. One study in the USA[5] found that providing an information leaflet did not decrease the number of hotel rooms that left lights on at night compared to when no leaflet was provided.

Background

Education programmes or awareness raising campaigns may be devised to address some of the threats posed to reptiles by humans. Such programmes may be aimed at a wide range of audiences, such as tourists, local residents, farmers or other businesses, and may tackle a wide range of threats, from exploitation through hunting, to the trade in exotic pets. In some cases, reptiles are protected by regulations and laws, but these may be difficult to enforce. Some infringements may be difficult to detect, whilst in other cases, people may be unaware of their responsibilities under such rules. Campaigns may be designed to increase compliance with laws, to encourage reporting of infringements (e.g. illegal hunting) or to reduce behaviours that can be a threat to reptiles (e.g. consumption of products derived from wild reptiles). Interventions may be carried out at a range of scales, from specific groups, largely through one-to-one interactions and targeted education programmes, through to large scale campaigns that spread awareness through broadcast and social media, signs and leaflets.

The effects of programmes may be measured in terms of the response of target species or in terms of changes in human behaviour that directly impact the magnitude of the threat.

A study in 1991–1992 on a sandy beach in Guanacaste Province, Costa Rica (1) found that while an education programme with local communities along with beach patrols for research and less frequently for turtle nest protection were taking place, there was a decrease in the percentage of leatherback turtle *Dermochelys coriacea* nests lost to poaching. Results were not statistically tested, and the effect of the different actions cannot be separated. In the first month of the education programme (October 1991), the percentage of nests lost to poaching was 91% (49 of 54 nests); in the second month (November) it was 51% (102 of 199 nests); and over the following four months it was 0–2% (of around 500 nests). In October–November 1991, an education and communications programme was carried out with local communities that involved organising trips to see the turtles, the chance to help with turtle research, lectures, lessons, slideshows, and local distribution of a brochure on leatherback turtle biology and conservation. Activities were also carried out with scout groups and the National Museum of Costa Rica (dates not provided). The beach was patrolled nightly for research purposes from October 1991–March 1992. Additional patrols were carried out by rural guards for three weeks in November and December, and periodically during January and February.

A study in 2002–2009 in mixed agricultural and forest habitats and human settlements in Kerala, India (2) found that educating the general public about snake identification, in particular the differences between venomous and non-venomous species, and creating a network of local snake experts lead to the prevention of a large number of non-venomous snakes being killed directly by humans. Results were not statistically tested. Over years following an education program and the creation of a network of local snake experts, local snake experts reported that they intervened to save 276 non-venomous snakes from being killed. The number of snakes that experts reported they saved from killing increased from 20 individuals in 2004 to 60 individuals in 2009. In 2002–2003, presentations about the reptiles in the region, the benefits of snakes, and how to identify them were given in 21 schools, five colleges and at least three villages. People reached by the program included 50 teachers, >400 students, 90 youth club members, and approximately 250 members of the general public. Participants were classed as local experts/citizen scientists (200 people) if they became actively involved in identifying snakes as part of the program, as well as monitoring and preventing snake kills in 2004–2009.

A before-and-after study in 1999–2010 in San Mariano municipality, Philippines (3) found that a communication, education, and public awareness campaign aimed at protecting the Philippine crocodile *Crocodylus mindorensis* resulted in the end of intentional crocodile killings by people and an increase in the crocodile population. Crocodile deaths caused by humans fell from 13 in 1998 to 0–1 in 2008–2010. The non-hatchling population in the municipality grew from 13 in 2002 to 64 in 2009. In addition, people reported crocodile nests to village officials rather than eating the eggs, and villages banned destructive fishing methods. In 1999, a project was set up to save crocodiles in an area, with communication outputs including billboards, wall paintings, posters, radio plugs, comic books, newsletters, school presentations, puppet shows, field visits and training workshops. The campaign focused on 15 villages, though the intensity of the campaign varied between villages.

A before-and-after study in 2008–2010 on coastal roads on the Caribbean Sea side of Dominica (4) found that running an awareness campaign and using road signs reduced lesser Antillean iguana *Iguana delicatissima* road mortality by half. After running an awareness campaign and putting up road signs to reduce driver speeds, lesser Antillean iguana road mortality reduced by 50% (0.3 fatal collisions/day) on coastal roads compared to beforehand (0.6 fatal collisions/day). An awareness campaign about protecting iguanas was carried out in May 2008–June 2010. The campaign included lectures at schools, presentations to government employees, radio and television interviews and distributing bumper stickers across the island asking people to slow down for iguanas. On 1 July 2009, road signs asking people to slow for iguanas were put up on coastal roads near known nesting locations (see original paper for details). Two coastal road segments (11–29 km long) were surveyed for iguanas every other day during the nesting season from April 2008–June 2010 (on 122 days before signs were put up and on 94 days afterwards).

A study in 2015–2016 in one hotel overlooking a sandy beach in Georgia, USA (5) found that when information leaflets detailing turtle-friendly behaviours were left in guest rooms, the number of lights visible from beach facing rooms was higher rather than lower in three of six months compared to when no leaflets were provided, and similar in the other three months. When the information leaflet was provided in 2015, light was visible from more rooms in May–July (59–98%), compared to when no leaflet was provided in May–July 2016 (39–64%). There was no significant difference in the number of rooms with visible lights in August–October (with leaflets in 2015: 40–71%; without leaflets in 2016: 50–64%). The information leaflet listed six options for turtle-friendly behaviour, one of which was to close curtains in beach-facing rooms and turn off outdoor lights. In May–October 2015, all guest rooms were provided with an information leaflet, and in May–October 2016 no leaflets were provided. In May–October in 2015–2016, counts of beach-facing rooms with visible lights were conducted three times/week, with surveys starting at 21:00 h. Data from 29 nights in 2015 and 29 in 2016 were included in the analysis.

A study in 2015–2016 in Saint Kitts (6) found that attending an educational summer camp about sea turtle conservation had mixed results on behaviour changes of attendees and their parents/guardians in relation to sea turtle conservation and turtle patrols. In response to a questionnaire, most former camp attendees and their parents/guardians reported a change in behaviour after attending the camp in relation to sea turtles (9–12-year-olds: 9 of 10; ≥ 13-years-olds: 19 of 24; parents/guardians: 34 of 39). Between 21 and 30% of attendees reported that they participated in a sea turtle patrol after attending the camp (9–12-year-olds: 3 of 10; ≥ 13-years-olds: 7 of 23; parents/guardians: 8 of 39). There was no significant increase in the likelihood of parents/guardians getting involved in marine conservation after their child attended the camp (data not reported). In 2007–2016, an annual sea turtle education camp was run that involved presentations, crafting activities and games on the topics of sea turtles and conservation. In 2015–2016, former camp attendees (attended 1–9 years previously) and their parents/guardians were invited to take part in the questionnaire regarding changes in their behaviour relating to sea turtles and the marine environment.

A replicated study in 2017 in 37 locations across six river drainage basins in northern Colombia (7) found that local residents exposed to turtle conservation initiatives claimed to have reduced their direct use of turtles compared to local residents not exposed to the initiatives, although stated rates of hunting, buying and selling of turtles remained similar. Fewer local residents exposed to conservation initiatives claimed to use the focal turtle species or other related turtle species as food (focal turtles: 10% of 50 participants; related turtles: 34% of 50) compared to local residents in areas with no conservation initiatives (focal turtles: 54% of 50 participants; sympatric turtles: 54% of 50). More local residents exposed to conservation initiatives claimed to have changed their consumption habits regarding focal turtle species (36% of 50 participants) compared to local residents not exposed to conservation initiatives (6% of 50 participants). However, stated rates of hunting, buying and selling turtles were similar whether or not residents had been exposed to conservation initiatives or not (see original paper for details). Semi-structured interviews with local residents were carried out in 37 locations that were classified into areas where turtle conservation initiatives had been implemented (17 locations, 50 survey participants) and areas where they had not (20 locations, 50 survey participants). Conservation initiatives included education in schools (1 initiative), community agreements to protect turtle habitat (2 initiatives), action against illegal wildlife trade (3 initiatives) and head-starting (12 initiatives).

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(2) Balakrishnan P. (2010) An education programme and establishment of a citizen scientist network to reduce killing of non-venomous snakes in Malappuram district, Kerala, India. *Conservation Evidence*, 7, 9–15.

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(7) Vallejo-Betancur M.M., Paez V.P. & Quan-Young L. (2018) Analysis of People's Perceptions of Turtle Conservation Effectiveness for the Magdalena River Turtle *Podocnemis lewyana* and the Colombian Slider *Trachemys callirostris* in Northern Colombia: An Ethnozoological Approach. *Tropical Conservation Science*, 11, 1–14.

15.2. Engage local communities in conservation activities

- **Six studies** evaluated the effects on reptile populations of engaging local communities in reptile conservation. One study was in each of the Philippines[1], Mozambique[2], Brazil[3], Costa Rica[4], Australia[5] and Colombia[6].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (5 STUDIES)

- **Abundance (1 study):** One site comparison study in Brazil[3] found that areas where community-based management of fishing practices was implemented had a higher abundance of river turtles than areas with no community-based management.
- **Reproductive success (3 studies):** Two before-and-after studies (including one site comparison study) in Mozambique[2] and Costa Rica[4] found that after involving the community in monitoring of nesting activity, fewer sea turtle eggs were lost to poaching than before projects began. One replicated, before-and-after study in Australia[5] found that when management of a saltwater crocodile egg harvest passed to an Indigenous management group, the number of eggs collected and hatching success of those eggs was lower than when it was run by an external company.
- **Survival (2 studies):** One study in the Philippines[1] found that after rural community members were paid a small incentive to protect Philippine crocodile sanctuaries combined with an education and awareness campaign, fewer crocodiles were killed than before community engagement. One before-and-after study in Mozambique[2] found that during a community-based turtle monitoring project no killing of adults was recorded.

BEHAVIOUR (0 STUDIES)

OTHER (1 STUDY)

- **Human behaviour change (1 study):** One replicated study in Colombia[6] found that in areas where communities were engaged in conservation initiatives relating to turtles, more people reported changing consumption habitats and fewer people reported using turtles for food compared to in areas with no initiatives, however, stated rates of hunting, buying and selling of turtles remained similar.

Background

When local community members are involved in managing local natural resources, they may have a greater interest in ensuring long-term sustainability of those resources. One potential outcome of this is a reduction in reptile persecution.

A study in 1999–2009 in freshwater and riparian zones in northern Luzon, Philippines (1) found that after rural community members were paid a small incentive to protect Philippine crocodiles *Crocodylus mindorensis* in three crocodile sanctuaries, along with being subject to an education and public awareness campaign, the number of crocodiles killed reduced. No Philippine crocodiles were killed in the sanctuaries between 2007 and 2009. The authors reported that most people in the area knew that crocodiles were legally protected. After a small population of crocodiles was discovered in 1999, three crocodile sanctuaries were created. The sanctuaries were protected by local community members who were paid a small incentive. A communication, education and public awareness campaign about the risks facing the crocodile was carried out (dates not provided) in the local rural communities. Details of monitoring and reporting of crocodile killings are not provided.

A before-and-after study in 2003–2007 on three beaches on Vamizi Island, Mozambique (2) found that a community-based sea turtle monitoring project appeared to reduce egg collection and hunting of adults. During the four years of a community turtle monitoring project, no egg collection (122 nests were laid/year on average) or hunting of female turtles was recorded. The authors reported that prior to the turtle monitoring project beginning, egg collection and hunting of adult female turtles was common within the local fishing community. Following the formation of two fishing village committees to manage local fishing resources and implement regulations, the committees created a turtle sanctuary around the north-east of the island to protect turtle breeding and feeding grounds. Three nesting beaches were monitored nightly for several months/year by 15 local turtle monitors supervised by a marine biologist in January–July 2003–2007.

A site comparison study in 2009 on a flood plain with mixed lakes and channels in Pará, Brazil (3) found that areas with community-based management (CBM) of fishing practices, including limiting use of gill-nets, seasonal fishing restrictions, protecting turtle nesting beaches and a ban on turtle trading, had more river turtles than areas without CBM. The effect of different aspects of the management programme cannot be separated. Overall, turtles (including *Podocnemis sextuberculata*, *Podocnemis unifilis* and *Podocnemis expansa*) were more abundant in areas with CBM (321 individuals) than in areas without CBM (33 individuals). *Podocnemis sextuberculata* abundance and biomass was higher in areas with CBM (14 individuals and 20 kg biomass/1,000 m^2 netting/12 hours) than in areas without (2 individuals and 3 kg biomass/1,000 m^2 netting/12 hours; data of other species not provided). The fishing agreement that formed the CBM programme had been in place for 20–30 years. While 13 communities in the area were a part of the fishing agreement, only two implemented the agreement. Turtle numbers were sampled at 14 sites (7 with CBM; 7 without CBM) in August–October 2009 using gill nets (15 nets/site; 215 m^2 nets; 3 each of 5 mesh sizes) with help from local fishers.

A before-and-after, site comparison study in 2005–2012 on a beach in Costa Rica (4) found that after involving the local community in monitoring olive ridley turtle *Lepidochelys olivacea* nests that were relocated to a hatchery and camouflaged, egg poaching decreased. Results were not statistically tested. Egg poaching reduced from 85% in the year before community monitoring began (2005) to 10% of eggs in 2006–2012. In 2006–2012, the local community was involved in monitoring turtle nesting activity and provided 24-hour monitoring to nests that were either relocated to an on-beach hatchery (363 nests, 38%nests) or camouflaged (595 nests, 62%; details of camouflaging method not provided) to discourage illegal collecting. Relocated nests were randomly allocated a 1 m^2 plot in the hatchery and dug into the sand. Hatchlings from both treatments were monitored on emergence and nests were excavated after hatching due dates to check hatching success.

A replicated, before-and-after study in 1989–2015 in freshwater swamps and tidal river banks within four river systems in Northern Territory, Australia (5) found that once an Indigenous management company took over the harvest and incubation of saltwater crocodile *Crocodylus porosus* eggs, hatching success rates reduced by more than a third. Results were not statistically tested. When a saltwater crocodile egg harvesting programme was under Indigenous management, incubation success rates were reduced (654 hatchlings from 1,396 live eggs/year) compared when it was under external management (1,413 hatchlings from 1,659 live eggs/year). Egg collection rates were also lower under Indigenous management (Indigenous management: 1,416 eggs harvested/year; external management: 2,359 eggs harvested/year). Saltwater crocodile eggs were collected and incubated as part of a regional government-led sustainable harvest initiative. In 1989–1997 an external management company ran the programme. In 1998–2015 it was run by a local Indigenous management company. In 1996–1997 eggs were harvested by the external company and incubated by the Indigenous management company. There was no harvest in 2007–2008. Annual quotas were 2,700–3,000 eggs/year (total limit of 70,000 eggs/year across the territory). Eggs were incubated at a constant temperature of 32°C and ≥99% humidity. Local workers were paid based on the number of eggs collected and hatchlings produced.

A replicated study in 2017 in 37 locations across six river drainage basins in northern Colombia (6) found that in areas where communities were engaged in conservation activities, local residents claimed to have reduced their direct use of turtles compared to local residents in areas that were not engaged, although stated rates of hunting, buying and selling of turtles remained similar. Fewer local residents in areas engaged in conservation initiatives claimed to use the focal turtle species or other related turtle species as food (focal turtles: 10% of 50 participants; related turtles: 34% of 50) compared to local residents in areas with no conservation initiatives (focal turtles: 54% of 50 participants; sympatric turtles: 54% of 50), and more claimed to have changed their consumption habits regarding focal turtle species (with conservation initiatives: 36% of 50 participants; without conservation initiatives: 6% of 50 participants). However, stated rates of hunting, buying and selling turtles were similar whether or not residents were in areas with conservation initiatives (see original paper for details). Semi-structured interviews with local residents were carried out in 37 locations that were classified into areas where turtle conservation initiatives had been implemented (17 locations, 50 survey participants) and areas where they had not (20 locations, 50 survey participants). Conservation initiatives included head-starting (12 initiatives), community agreements to protect turtle habitat (2 initiatives), action against illegal wildlife trade (3 initiatives) and education in schools (1 initiative).

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(2) Garnier J., Hill N., Guissamulo A., Silva I., Witt M. & Godley B. (2012) Status and community-based conservation of marine turtles in the northern Querimbas Islands (Mozambique). *Oryx*, 46, 359–367.

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(4) James R. & Melero D. (2015) Nesting and conservation of the Olive Ridley sea turtle (*Lepidochelys olivacea*) in playa Drake, Osa Peninsula, Costa Rica (2006–2012). *Revista De Biologia Tropical*, 63, 117–129.

(5) Corey B., Webb G.J.W., Manolis S.C., Fordham A., Austin B.J., Fukuda Y., Nicholls D. & Saalfeld K. (2018) Commercial harvests of saltwater crocodile *Crocodylus porosus* eggs by Indigenous people in northern Australia: Lessons for long-term viability and management. *Oryx*, 52, 697–708.

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15.3. Engage policy makers to make policy changes beneficial to reptiles

- We found no studies that evaluated the effects on reptile populations of engaging policy makers to make policy changes beneficial to reptiles.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Engaging with and raising awareness amongst policymakers (e.g. CITES management and scientific authorities) about specific threats to reptiles, and the need for conservation, may result in improved legal protection of reptiles and their habitats.

15.4. Offer reptile-related eco-tourism to improve behaviour towards reptiles

- **Two studies** evaluated the effects on reptile populations of offering reptile-related eco-tourism to improve behaviour towards reptiles. One study was in the USA[1] and one was in St Kitts and Nevis[2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (0 STUDIES)

OTHER (1 STUDY)

- **Human behaviour change (1 study):** One study in the USA[1] reported that 32% of respondents to a survey said they would have gone to look for a nesting turtle if they had not been able to join a supervised turtle watch. One study in St Kitts and Nevis[2] found that people who attended a leatherback turtle tour reported that they would be more conscientious of how their behaviours on the beach affected sea turtles.

Background

Large numbers of tourists visiting important reptile habitats may pose a threat to wild reptile populations. Eco-tourism may help to promote reptile conservation and raise funds for conservation and research, as well as providing an opportunity for tourists to observe and experience wild reptiles while keeping disturbance to a minimum. However, where eco-tourism is not carefully regulated, increased levels of human disturbance may end up being detrimental to reptile populations (Iverson *et al.* 2006).

Iverson J.B., Converse S.J., Smith G.R. & Valiulis J.M. (2006) Long-term trends in the demography of the Allen Cays Rock Iguana (*Cyclura cychlura inornata*): Human disturbance and density-dependent effects. *Biological Conservation*, 132, 300–310.

A replicated study in 1994 involving six organizations running loggerhead turtle *Caretta caretta* watching trips in Florida, USA (1) reported that a third of tourists reported that they would have tried to observe a nesting turtle independently if the opportunity to join a supervised turtle watch was not available. In response to a questionnaire, 32% of respondents said that they would have gone to look for a nesting turtle if they had not been able to join a supervised turtle watch. Loggerhead turtle watches were carried out by six organizations that had been issued permits by the Florida Department of Environmental Protection. In 1994, these organisations distributed questionnaires to members of the public that attended turtles watches and were over the age of 15. Of the 1,148 questionnaires given out, 608 were returned and 488 were completed correctly and included in the analysis.

A study in 2009–2014 on St Kitts, St Kitts and Nevis (2) found that people who attended a leatherback turtle *Dermochelys coriacea* tour reported that after going on the tour they would be more conscientious of how their behaviours on the beach affected sea turtles. All 38 people that responded to survey after attending a tour reported that in future, they would be more conscientious of how their behaviours on the beach affected sea turtles. Thirty-six (97%) respondents also reported that they would be more likely to report sightings of turtle nests or injured turtles. In 2009–2014, leatherback turtle ecotours were carried out during

April–June. Tours involved a maximum of 10 people visiting a turtle nesting beach with a trained guide to observe nesting turtles. Attendees also received a briefing and education material on sea turtles. In 2014, a survey was distributed to 206 attendees of the tour that had provided contact information.

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15.5. Provide training for local staff in species identification

- We found no studies that evaluated the effects on reptile populations of providing training for local staff in species identification.

'We found no studies' means that we have not yet found any studies that have directly evaluated this intervention during our systematic journal and report searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Species identification skills play an important role in recognising the conservation needs of reptile populations, as well as the threats some reptiles may pose to humans. Where staff across a range if sectors (e.g. government agencies, building and development sector, conservation organisations) have good identification skills they may be better able to respond to situations involving reptiles with appropriate and proportionate actions, with benefits for wild reptile populations.

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Appendix 1: Journals (and years) searched

a) Specialist reptile journals (and years) for which new (17 journals) or updated (11 journals) searches were carried out by the authors of this synopsis and summaries were completed as a result of searches.

A total of 33 specialist reptile English journals were searched. Asterisk indicates updated searches. Bold indicates new journal searches for this synopsis.

Journal	Years searched
Acta Herpetologica*	2006–2018
African Journal of Herpetology (formerly The Journal of the Herpetological Association of Africa)*	1990–2018
Amphibia-Reptilia*	1980–2018
Amphibian and Reptile Conservation*	1996–2018
Applied Herpetology	2003–2009
Asian Herpetological Research	2010–2018
Asiatic Herpetological Research	1993–2008
Basic and Applied Herpetology	2011–2018
Bibliotheca Herpetologica	1999–2017
Bulletin of the Herpetological Society of Japan	1999–2008
Bulletin of the Maryland Herpetological Society	1980–2015
Caribbean Herpetology	2010–2018
Chelonian Conservation and Biology	1993–1996, 2005–2018
Chelonian Research Monographs	1996–2017
Collinsorum (formerly Journal of Kansas Herpetology)	2002–2018
Contemporary Herpetology	1998–2009
Copeia*	1910–2018
Current Herpetology (formerly Acta Herpetologica Japonica 1964–1971 and Japanese Journal of Herpetology 1972–1999)*	1964–2018
Herpetologica*	1936–2018
Herpetological Conservation and Biology*	2006–2018
Herpetological Monographs*	1982–2018
Herpetological Review	1967–2018
Herpetology Notes	2008–2018
Herpetozoa	1988–2018
Journal of Herpetology*	1968–2018
Journal of North American Herpetology	2014–2017
Kansas Herpetological Society Newsletter	1974–2001
Mesoamerican Herpetology	2014–2017
Phyllomedusa	2002–2018
South American Journal of Herpetology*	2006–2018
Testudo	1978–2017
The Herpetological Bulletin*	2008–2018

The Herpetological Journal*	2002–2016

b) All other journals (and years) searched for the discipline-wide Conservation Evidence database for which summaries were completed as a result of the searches (264)

An asterisk indicates the journals most relevant to this synopsis.

Journal	Years searched	Topic
Acrocephalus	2009–2018	All biodiversity
Acta Chiropterologica	1999–2018	All biodiversity
Acta Oecologica	1990–2018	All biodiversity
African Bird Club Bulletin	1994–2017	All biodiversity
African Journal of Ecology	1963–2016	All biodiversity
African Journal of Marine Science	1983–2018	All biodiversity
African Primates	1995–2012	All biodiversity
African Zoology	1979–2013	All biodiversity
Agriculture, Ecosystems and Environment*	1983–2017	All biodiversity
Agroforestry Systems (Springer)	1982–2007	All biodiversity
Ambio	1972–2011	All biodiversity
American Journal of Primatology	1981–2014	All biodiversity
American Naturalist	1867–2018	All biodiversity
Animal Biology	2003–2013	All biodiversity
Animal Conservation*	1998–2018	All biodiversity
Animal Welfare	1992–2016	All biodiversity
Annales Zoologici Fennici	1964–2013	All biodiversity
Annales Zoologici Societatis Zoologicae Botanicae Fennicae Vanamo	1932–1964	All biodiversity
Annual Review of Ecology, Evolution, and Systematics (formerly Annual Review of Ecology and systematics 1970-2002)*	1970–2018	All biodiversity
Antarctic Science	1980–2018	All biodiversity
Anthrozoos	1987–2013	All biodiversity
Apidologie	1958–2009	All biodiversity
Applied Animal Behaviour Science	1988–2014	All biodiversity
Applied Vegetation Science	1998–2017	All biodiversity
Aquatic Biology	2007–2018	All biodiversity
Aquatic Botany	1975–2017	All biodiversity
Aquatic Conservation: Marine and Freshwater Ecosystems	1991–2018	All biodiversity
Aquatic Ecology (Springer)	1968–2018	All biodiversity
Aquatic Ecosystem Health & Management	1998–2018	All biodiversity
Aquatic Invasions	2006–2016	All biodiversity
Aquatic Living Resources (Ressources Vivantes Aquatiques)	1988–2018	All biodiversity
Aquatic Mammals	1972–2018	All biodiversity

Ardeola	1996–2018	All biodiversity
Arid Land Research and Management (formerly Arid Soil Research and Rehabilitation (1987 - 2000)	1987–2013	All biodiversity
Asian Primates	2008–2012	All biodiversity
Auk	1980–2016	All biodiversity
Austral Ecology	1977–2018	All biodiversity
Australian Mammalogy	2000–2018	All biodiversity
Avian Conservation and Ecology	2005–2016	All biodiversity
Basic and Applied Ecology*	2000–2018	All biodiversity
Behavioral Ecology	1990-2013	All biodiversity
Behaviour	1948-2013	All biodiversity
Biocontrol (formerly Entomophaga until 1998)	1956–2016	All biodiversity
Biocontrol Science and Technology	1991–2016	All biodiversity
Biodiversity and Conservation*	1994–2018	All biodiversity
Biological Conservation* (Elsevier)	1981–2018	All biodiversity
Biological Control	1991–2017	All biodiversity
Biological Invasions	1999–2017	All biodiversity
Biology and Environment: Proceedings of the Royal Irish Academy	1993–2017	All biodiversity
Biology Letters	2005–2018	All biodiversity
Biotropica	1990–2018	All biodiversity
Bird Conservation International	1991–2016	All biodiversity
Bird Study	1980–2016	All biodiversity
Boreal Environment Research	1996–2014	All biodiversity
Bulletin Français de la Pêche et de la Pisciculture	1986–2007	All biodiversity
Canadian Journal of Fisheries and Aquatic Sciences	1901–2018	All biodiversity
Canadian Journal of Forest Research	1971–2013	All biodiversity
Caribbean Journal of Science	1961–2013	All biodiversity
CCAMLR Science	1985–2016	All biodiversity
CEE (Collaboration for Environmental Evidence) Systematic Reviews	2004–2017	All biodiversity
Coastal Engineering	2000–2018	All biodiversity
Community Ecology	2000–2012	All biodiversity
Conservation Biology*	1987–2018	All biodiversity
Conservation Evidence*	2004–2018	All biodiversity
Conservation Genetics	2000–2013	All biodiversity
Conservation Letters*	2008–2018	All biodiversity
Contributions to Primatology	1974–1991	All biodiversity
Cunninghamia	1981–2016	All biodiversity
Dodo	1977–2001	All biodiversity
Ecological and Environmental Anthropology	2005–2008	All biodiversity
Ecological Applications*	1991–2018	All biodiversity
Ecological Entomology	1985–2018–	All biodiversity
Ecological Indicators	2001–2007	All biodiversity

Ecological Management and Restoration	2000–2018	All biodiversity
Ecological Restoration*	1981–2018	All biodiversity
Ecology*	1936–2018	All biodiversity
Ecology Letters	1998–2013	All biodiversity
Écoscience	1994–2013	All biodiversity
Ecosystems	1998–2013	All biodiversity
Emu	1980–2016	All biodiversity
Endangered Species Bulletin	1966–2003	All biodiversity
Endangered Species Research	2004–2017	All biodiversity
Entomologia Experimentalis et Applicata	2015–2018	All biodiversity
Environmental Conservation*	1974–2018	All biodiversity
Environmental Entomology	1990–2018	All biodiversity
Environmental Evidence*	2012–2018	All biodiversity
Environmental Management*	1977–2018	All biodiversity
Environmentalist	1981–1988	All biodiversity
Estuaries and Coasts	2013–2017	All biodiversity
Ethology Ecology & Evolution	1989–2014	All biodiversity
European Journal of Soil Science	1950–2012	Soil fertility
European Journal of Wildlife Research*	2004–2018	All biodiversity
Evolutionary Anthropology	1992–2014	All biodiversity
Evolutionary Ecology	1987–2014	All biodiversity
Evolutionary Ecology Research	1999–2014	All biodiversity
Fire Ecology	2005–2016	All biodiversity
Fish and Fisheries	2000–2018	All biodiversity
Fisheries	2017–2018	All biodiversity
Fisheries Management and Ecology	1994–2018	All biodiversity
Fisheries Oceanography	1992–2018	All biodiversity
Fisheries Research	1990–2018	All biodiversity
Flora	1991–2017	All biodiversity
Folia Primatologica	1963–2014	All biodiversity
Folia Zoologica	1959–2013	All biodiversity
Forest Ecology and Management	1976–2018	All biodiversity
Freshwater Biology	1975–2016	All biodiversity
Freshwater Science (formerly Freshwater Invertebrate Biology 1982-1985 and Journal of the North American Benthological Society 1986-2011)	1982–2018	All biodiversity
Frontiers in Marine Science	2017–2018	All biodiversity
Functional Ecology	1987–2013	All biodiversity
Genetics and Molecular Research	2002–2013	All biodiversity
Geoderma	1967–2012	Soil fertility
Gibbon Journal	2005–2011	All biodiversity
Global Change Biology	1995–2017	All biodiversity
Global Ecology and Biogeography	1991–2014	All biodiversity
Global Ecology and Conservation	2014–2018	All biodiversity
Grass and Forage Science	1980–2017	All biodiversity

Human Wildlife Interactions (formerly Human Wildlife Conflicts)*	2007–2017	All biodiversity
Hydrobiologia	2000–2018	All biodiversity
Hystrix, the Italian Journal of Mammalogy	1994–2018	All biodiversity
Ibis	1980–2016	All biodiversity
ICES Journal of Marine Science	1990–2018	All biodiversity
iForest~~s~~	2008–2016	All biodiversity
Insect Conservation and Diversity	2008–2018	All biodiversity
Integrative Zoology	2006–2013	All biodiversity
International Journal of Pest Management (formerly PANS Pest Articles & News Summaries 1969 - 1975, PANS 1976-1979 & Tropical Pest Management 1980-1992)	1969–1979	All biodiversity
International Journal of Primatology (Springer)	1980–2012	All biodiversity
International Journal of the Commons	2007–2016	All biodiversity
International Journal of Wildland Fire	1991–2016	All biodiversity
International Wader Studies	1970–1972	All biodiversity
International Zoo Yearbook	1960–2015	All biodiversity
Invasive Plant Science and Management	2008–2016	All biodiversity
Israel Journal of Ecology & Evolution	1963–2013	All biodiversity
Italian Journal of Zoology	1978–2013	All biodiversity
Journal for Nature Conservation (English 2002-)*	2002–2018	All biodiversity
Journal of Animal Ecology (BES)*	1932–2018	All biodiversity
Journal of Apicultural Research	1962–2009	All biodiversity
Journal of Applied Ecology*	1964–2018	All biodiversity
Journal of Aquatic Plant Management (formerly Hyacinth Control Journal)	1962–2016	All biodiversity
Journal of Arid Environments	1993–2017	All biodiversity
Journal of Avian Biology (formerly Ornis Scandinavica 1970 – 1993)	1980–2016	All biodiversity
Journal of Bat Research & Conservation	2000–2018	All biodiversity
Journal of Cetacean Research and Management	1999–2018	All biodiversity
Journal of Coastal Research	2015–2018	All biodiversity
Journal of Ecology*	1933–2018	All biodiversity
Journal of Environmental Management*	1973–2018	All biodiversity
Journal of Experimental Marine Biology & Ecology	2000–2012	All biodiversity
Journal of Field Ornithology	1980–2016	All biodiversity
Journal of Forest Research	1996–2018	All biodiversity
Journal of Great Lakes Research	1975–2017	All biodiversity
Journal of Insect Conservation	1997–2018	All biodiversity
Journal of Insect Science	2003–2018	All biodiversity
Journal of Mammalian Evolution	1993–2014	All biodiversity
Journal of Mammalogy	1919–2018	All biodiversity
Journal of Mountain Science	2004–2016	All biodiversity

Journal of Negative Results: Ecology and Evolutionary Biology	2004–2016	All biodiversity
Journal of Ornithology	2004–2018	All biodiversity
Journal of Primatology	2012–2013	All biodiversity
Journal of Raptor Research	1966–2016	All biodiversity
Journal of Sea Research (formerly Netherlands Journal of Sea Research)	1961–2018	All biodiversity
Journal of the Marine Biological Association of the United Kingdom	1887–2018	All biodiversity
Journal of Threatened Taxa	2009–2013	Plant conservation
Journal of Tropical Ecology*	1986–2018	All biodiversity
Journal of Vegetation Science	1990–2017	All biodiversity
Journal of Wetlands Ecology	2008–2012	All biodiversity
Journal of Wetlands Environmental Management	2012–2016	All biodiversity
Journal of Wildlife Diseases	1965–2012	All biodiversity
Journal of Zoology*	1966–2018	All biodiversity
Jurnal Primatologi Indonesia	2009	All biodiversity
Knowledge and Management of Aquatic Ecosystems (formerly Bulletin Français de la Pêche et de la Pisciculture)	2008–2018	All biodiversity
Lake and Reservoir Management	1984–2016	All biodiversity
Land Degradation and Development	1989–2016	All biodiversity
Land Use Policy	1984–2012	Soil fertility
Latin American Journal of Aquatic Mammals	2002–2018	All biodiversity
Lemur News	1993–2012	All biodiversity
Limnologica - Ecology and Management of Inland Waters	1999–2018	All biodiversity
Mammal Research (formerly Acta Theriologica until 2000)	1977–2017	All biodiversity
Mammal Review	1970–2012;2013–2018	Bats; All biodiversity
Mammal Study	2005–2018	All biodiversity
Mammalia	1937–2018	All biodiversity
Mammalian Biology	2002–2018	All biodiversity
Mammalian Genome	1991–2013	All biodiversity
Management of Biological Invasions	2010–2016	All biodiversity
Mangroves and Saltmarshes (Springer)	1996–1999	All biodiversity
Marine and Freshwater Research	1980–2018	All biodiversity
Marine Ecology	1980–2018	All biodiversity
Marine Ecology Progress Series	2000–2018	All biodiversity
Marine Environmental Research	1978–2018	All biodiversity
Marine Mammal Science	1985–2018	All biodiversity
Marine Pollution Bulletin	2010–2018	All biodiversity
Mires and Peat	2006–2016	All biodiversity

Natural Areas Journal	1992–2017	All biodiversity
Nature Conservation	2012–2018	All biodiversity
NeoBiota	2011–2017	All biodiversity
Neotropical Entomology	2004–2018	All biodiversity
Neotropical Primates	1993–2012	All biodiversity
New Journal of Botany	2011–2013	All biodiversity
New Zealand Journal of Marine and Freshwater Research	1967–2018	All biodiversity
New Zealand Journal of Zoology*	1974–2018	All biodiversity
New Zealand Plant Protection	2000–2016	All biodiversity
Northwest Science	2007–2016	All biodiversity
Oecologia*	1969–2018	All biodiversity
Oikos*	1949–2018	All biodiversity
Ornitologia Neotropical	1990–2018	All biodiversity
Oryx*	1950–2018	All biodiversity
Ostrich	1980–2016	All biodiversity
Pacific Conservation Biology*	1993–2018	All biodiversity
Pakistan Journal of Zoology	2004–2013	All biodiversity
Plant Ecology	1948–2007	All biodiversity
Plant Protection Quarterly	2008–2016	All biodiversity
Polish Journal of Ecology	2002–2013	All biodiversity
Population Ecology	1952–2013	All biodiversity
Preslia	1973–2017	All biodiversity
Primate Conservation	1981–2014	All biodiversity
Primates	1957–2013	All biodiversity
Rangeland Ecology & Management (previously Journal of Range Management 1948 -2004)	1948-2016	All biodiversity
Raptors Conservation	2005–2016	All biodiversity
Regional Studies in Marine Science	2015–2018	All biodiversity
Restoration Ecology*	1993–2018	All biodiversity
Revista de Biologia Tropical	1976–2018	All biodiversity
Riparian Ecology and Conservation	2013–2017	All biodiversity
River Research and Applications	1987–2016	All biodiversity
Russian Journal of Ecology	1993–2013	All biodiversity
Slovak Raptor Journal	2007–2016	All biodiversity
Small Ruminant Research	1988–2017	All biodiversity
Soil Biology and Biochemistry	1969–2012	Soil fertility
Soil Use and Management	1985–2012	Soil fertility
South African Journal of Botany	1982–2016	All biodiversity
South African Journal of Wildlife Research	1971–2014	All biodiversity
Southern Forests	2008–2013	All biodiversity
Systematic Reviews Centre for Evidence-Based Conservation*	2004–2017	All biodiversity
The Canadian Field-Naturalist (formerly Ottawa Naturalist)	1987–2018	All biodiversity
The Condor	1980–2016	All biodiversity

The Journal of Wildlife Management*	1945–2018	All biodiversity
The Open Ornithology Journal	2008–2016	All biodiversity
The Rangeland Journal	1976–2016	All biodiversity
The Southwestern Naturalist	1956–2018	All biodiversity
The Wilson Journal of Ornithology (formerly The Wilson Bulletin)	1980–2016	All biodiversity
Trends in Ecology and Evolution*	1986–2017	All biodiversity
Tropical Conservation Science	2008–2018	All biodiversity
Tropical Ecology	1960–2018	All biodiversity
Tropical Grasslands	1967–2010	All biodiversity
Tropical Zoology	1988–2018	All biodiversity
Turkish Journal of Zoology	1996–2014	All biodiversity
Vietnamese Journal of Primatology	2007–2009	All biodiversity
Wader Study Group Bulletin	1970–1977	All biodiversity
Waterbirds (formerly Colonial Waterbirds)	1983–2016	All biodiversity
Weed Biology and Management	2001–2016	All biodiversity
Weed Research	1961–2017	All biodiversity
West African Journal of Applied Ecology	2000–2016	All biodiversity
Western North American Naturalist	2000–2017	All biodiversity
Wetlands	1981–2016	All biodiversity
Wetlands Ecology and Management [also see Mangroves and Saltmarshes (Springer)]	1989–2016	All biodiversity
Wildfowl	1969–2018	All biodiversity
Wildlife Biology	1995–2013	All biodiversity
Wildlife Monographs	1958–2013	All biodiversity
Wildlife Research (CSIRO publishing) (formerly CSIRO Wildlife Research until 1973)	1974–2018	All biodiversity
Wildlife Society Bulletin	1973–2018	All biodiversity
Zhurnal Obshchei Biologii	1972–2013	All biodiversity
Zoo Biology	1982–2016	All biodiversity
Zookeys	2008–2013	All biodiversity
Zoologica Scripta	1971–2014	All biodiversity
Zoological Journal of the Linnean Society	1856–2013	All biodiversity
Zootaxa	2004–2014	All biodiversity

Appendix 2: Conservation reports (and years) searched

Conservation reports published by a total of 7 organisations were searched.

a) New searches for this synopsis

Organization	Years searched	Details
IUCN-SSC Crocodile Specialist Group	2006–2018	CSG Articles
IUCN SSC Crocodile Specialist Group	2005–2017	CSG Reports

b) All other conservation reports searched for the discipline-wide Conservation Evidence database

An asterisk indicates the reports most relevant to this synopsis.

Organization	Years searched	Details
Amphibian Survival Alliance	1994–2012	Vol 9–Vol 104
British Trust for Ornithology	1981–2016	Report 1–687
IUCN-SSC Invasive Species Specialist Group	1995–2013	Aliens: The Invasive Species Bulletin (IUCN) Vol 1–33
Joint Nature Conservation Committee*	1991–2018	Reports 1–627
Natural England*	1991–2018	
NatureScot*	2004–2018	Reports 1–945

Appendix 3: Literature reviewed for the Reptile Synopsis

The diagram below shows the total numbers of journals and report series searched for this synopsis, the total number of publications searched (title and abstract) within those, and the number of publications that were summarized from each source of literature.