Subtidal Benthic Invertebrate Conservation

Global evidence for the effects of interventions

Anaëlle J. Lemasson, Laura R. Pettit,
Rebecca K. Smith & William J. Sutherland

SYNOPSES OF CONSERVATION EVIDENCE SERIES

Subtidal Benthic Invertebrate Conservation

Global evidence for the effects of interventions

Anaëlle J. Lemasson, Laura R. Pettit, Rebecca K. Smith and William J. Sutherland

Synopses of Conservation Evidence

Lemasson, A.J., Pettit, L.R., Smith, R.K., and Sutherland, W.J. (2019) *Subtidal Benthic Invertebrate Conservation: Global Evidence for the Effects of Interventions.* Synopses of Conservation Evidence Series. University of Cambridge, Cambridge, UK.

Cover image: Black brittle stars *Ophiocomina nigra* on sand covered rock in the Solan Bank Reef Special Area of Conservation, North-East Atlantic Ocean, UK. [EUNIS level 4 A4.21 Echinoderms and crustose communities on circalittoral rock. Annex 1 Reef]. All rights reserved. ©JNCC.

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

Contents

Advisory Board

We thank the following people for advising on the scope and content of this synopsis:

Dr. Lara Atkinson	South African Environmental Observation Network/University of Cape Town, South Africa
Prof. Martin Attrill	University of Plymouth, UK
Prof. Marnie Campbell	Murdoch University, Australia
Prof. Tasman Crowe	University College Dublin, Ireland
Prof. Qinhua Fang	Xiamen University, China
Prof. Simonetta Fraschetti	Università di Napoli Federico II, Italy
Dr. Joaquim Garrabou	Institut de Ciències del Mar, Spain
Dr. Alan Jordan	New South Wales Department of Primary Industries, Australia
Dr. Ian McLeod	James Cook University, Australia
Prof. Marco Milazzo	Universita degli Studi di Palermo, Italy
Dr. Maite Narvarte	Universidad Nacional del Comahue, Argentina
Dr. Jenny Oates	World Wildlife Fund, UK
Dr. Mark Spalding	The Nature Conservancy, Italy
Dr. Tim Stevens	Griffith University, Australia
Dr. Karen Webb	Joint Nature Conservation Committee, UK
Prof. Steven Widdicombe	Plymouth Marine Laboratory, UK

About the authors

Dr. Anaëlle J. Lemasson is a post-doctoral Research Associate for the Joint Nature Conservation Committee, UK, and specialist in marine biology and ecology.

Dr. Laura R. Pettit is a Senior Habitats Adviser for the Joint Nature Conservation Committee, UK.

Dr. Rebecca K. Smith is a Senior Research Associate in the Department of Zoology, University of Cambridge, UK.

Professor William J. Sutherland is the Miriam Rothschild Professor of Conservation Biology at the University of Cambridge, UK.

Acknowledgements

This synopsis project was possible with funding from the Joint Nature Conservation Committee (JNCC).

Thanks go to Yessica Griffith (JNCC) for her guidance and support throughout the conception of this synopsis, to the JNCC Marine Management Team for their advice on offshore industries, to Leo Clarke (Bangor University) and Natasha Taylor, Chris Barrett, Khatija Alliji and Ross McIntyre (Cefas) for their willingness to share their fisheries knowledge, and finally to the team at Conservation Evidence, Cambridge, for their expert advice and guidance.

1. About this book

1.1 The Conservation Evidence project

The Conservation Evidence project is constituted of 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 that was identified. 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 5,500 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 the synopses. This is available as part of the searchable database and is published as an updated book edition each year (https://www.conservationevidence.com/content/page/79).

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

You can learn more about the Conservation Evidence project and the methods behind it in Sutherland *et al.* (2019).

1.2 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 5,500 studies so far) on the effects of interventions to conserve and restore 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	• Weight or numerically evaluate the evidence according to its quality
• Work in partnership with conservation practitioners, policymakers, and scientists to develop the list of interventions and ensure	• Provide recommendations for conservation problems, but instead

we have covered the most important literature	provide scientific information to help with decision-making

1.3 Who is this synopsis for?

If you are reading this, we hope you are someone who has to or wants to make decisions about how best to support, manage, and conserve the marine environment and its biodiversity. You might be a marine conservationist in the public or private sector, a campaigner, a marine advisor or consultant, a policymaker, a researcher, someone taking action to protect the marine environment, or a concerned citizen. This synopsis summarizes 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 or others' planned actions could have. Here, by "evidence", we mean any scientific studies found during our systematic searches (see below section 1.6) that quantitatively report the effects of conservation actions (interventions).

When decisions have to be made with particularly important or irreversible 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).

1.4 Background

It is now widely recognised that the marine environment is highly biodiverse, and that this abundant biodiversity is key to the provision of essential goods and services to humans, and to human well-being (Garmfeldt *et al*. 2015). However, marine biodiversity is facing multiple threats from impactful human activities and human-induced climate change (Lotze *et al*. 2018). There is therefore an increasing need for evidence-based management and conservation of the marine environment and of all organisms that live in it.

As such, policy makers and managers need to assess the impacts of these pressures on the marine environment and to recommend and implement measures that restrain, reduce or eliminate these pressures and impacts. These activities are undertaken by multi-disciplinary organisations, including academic institutions, international, governmental and regulatory agencies, devolved governments, local authorities, non-governmental organisations, and science advisors. When assessing potential pressures on the marine environment, each of these bodies employs staff to scrutinise the available scientific evidence-base for guidance on best practice to reduce impacts.

Reviewing the evidence is a time-consuming and costly exercise. While a large amount of evidence exists, it is often not collated and summarised in an easily accessible format. In addition, in general, the assessment of the evidence-base is approached on a case-by-case basis and different stakeholders independently conduct evidence reviews relative to their specific application or enquiry. This approach is counter to the philosophy of 'produce once and use many times over' and is a highly inefficient use of resources. This synopsis summarises the available global scientific evidence of the effectiveness of conservation interventions for subtidal benthic invertebrate populations, including management measures for impactful resource extracting activities that take place in the marine environment (e.g. offshore industries). The methods used to create it are outlined below and are designed to efficiently do so without multiplying effort and resources. The output of this synopsis contributes to the maintenance and enhancement of marine biodiversity and associated environmental resources.

1.5 Scope of the review

1.5.1 Review subject

This synopsis covers published evidence for the effects of global conservation interventions, and more generally management interventions, aimed primarily at conserving, but also at restoring and promoting, subtidal benthic invertebrate species and communities. This includes invertebrates using all seabed habitats which are permanently covered by seawater, apart from coral species (deep-sea and tropical). Evidence for the effects of interventions on coral species have not been included in this synopsis due to the large amount of literature which examines their conservation; but they will be covered separately. However, actions aimed to conserve or restore non-invertebrate species or coral species have been included where these species create biogenic habitats that can be inhabited by invertebrate species. Such species include, for instance, seagrass and eelgrass species which form seagrass meadows, kelp species which form kelp forests, mangrove species which form mangrove forests, and corals species which form coral reefs. In these instances, studies presenting evidence for the conservation of these habitat-forming species have been included, but only data related to the associated invertebrate species (except coral) have been reported, while data on the habitat-forming species (seagrass, kelp, mangrove, coral) will be the focus of different synopses.

The present synthesis, focussing on evidence for the effectiveness of global interventions for the conservation of subtidal benthic invertebrates, has not yet been covered using subject-wide evidence synthesis. This is defined as a systematic method of evidence synthesis that covers entire subjects 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). The topic is therefore a priority for the discipline-wide Conservation Evidence database. To help with the sometimes-complex vocabulary used to describe the marine environment, and for which a plain English equivalent to not exist, we provide a Glossary of terms (Appendix 1).

This synthesis covers evidence for the effects of conservation interventions for wild subtidal benthic invertebrates (i.e. not in captivity). We did not include evidence from the substantial literature on husbandry of commercially reared cultured marine invertebrates or those kept in zoos. However, where these interventions are relevant to the conservation of wild declining or threatened species, they were included, e.g. captive breeding (such as shellfish hatcheries) for the purpose of reintroductions, transplantation, stock enhancement, or gene banking (for future release). This global synthesis collates evidence for the effects of conservation actions for all subtidal benthic invertebrates (except coral species) across all marine habitats.

1.5.2. Advisory board

An advisory board made up of international conservationists and academics with expertise in seabed management and marine invertebrate conservation was formed. These experts provided input into the evidence synthesis at two key stages: a) developing a comprehensive list of conservation interventions for review and b) reviewing the draft evidence synthesis. The advisory board is listed above.

1.5.3. Creating the list of interventions

At the start of the project, a comprehensive list of interventions was developed by scanning 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 chosen by Conservation Evidence (described below). The aim was to include all actions that have been carried out or advised to support populations or communities of wild marine subtidal benthic invertebrates, whether evidence for the effectiveness of an action is available or not. During the synthesis process further interventions were discovered and integrated into the synopsis structure.

The list of interventions was organized into categories based on the International Union for the Conservation of Nature (IUCN) classifications of direct threats (http://www.iucnredlist.org/technical-documents/classification-schemes/threats-classification-scheme) and conservation actions (http://www.iucnredlist.org/technical-documents/classification-schemes/conservation-actions-classification-scheme-ver2).

In total, we found 226 conservation and/or management interventions that could be carried out to conserve subtidal benthic invertebrate populations. We found evidence for the effects on subtidal benthic invertebrate populations of 85 of these interventions. The evidence was reported as 239 summaries from 204 relevant publications found during our searches (see Methods below).

1.6 Methods

1.6.1 Literature searches

Literature was obtained from the Conservation Evidence discipline-wide literature database, and from searches of additional subject specific literature sources (see Appendices 2 & 3). The Conservation Evidence discipline-wide literature database is compiled using systematic searches of journals; relevant publications that describe studies of conservation interventions for all species groups and habitats are saved from each journal and are added to the database. The final list of evidence sources searched for this synopsis is published in this synopsis document – see Appendix 2, and the full list of journals and report series searched is published online (https://www.conservationevidence.com/journalsearcher/synopsis).

a) Global evidence

Evidence from all around the world was included.

b) Languages included

Only English language journals were included in this synopsis. A recent study on the topic of language barriers in global science indicates that approximately 35% of conservation studies may be in non-English languages (Amano et al. 2016). While searching only English language journals may therefore potentially introduce some bias to the review process, project resources and time constraints determined the number of journals that could be searched within the project timeframe.

b) Journals searched

i) From Conservation Evidence discipline-wide literature database

All of the journals (and years) listed in Appendix 2b were searched prior to or during the completion of this project by authors of other synopses, 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 include papers relevant to this synopsis, but if they did, those papers were summarised.

ii) Update searches

The authors of this synopsis updated the search of *Hydrobiologia* for the year 2017 (Appendix 2). For the year 2017, searches of other journals previously searched by Conservation Evidence were updated by authors of other synopses.

iii) New searches

Additional, focussed searches of journals most relevant to the conservation of subtidal benthic invertebrate populations listed in Appendix 2a were undertaken. 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. Due to time constraints, some of the journals listed below were not systematically searched using the standard Conservation Evidence methodology of subject-wide evidence synthesis (Sutherland *et al.* 2019), but instead using strings of keywords (⧗ means that searches were done <u>only</u> using keywords– see Appendix 2a).

- African Journal of Marine Science
- Aquatic Conservation: Marine and Freshwater Ecosystems
- Estuarine, Coastal and Shelf Science⧗
- Fisheries Research⧗
- Marine Environmental Research
- Regional Studies in Marine Science
- Journal of Sea Research (formerly known as Netherlands Journal of Sea Research)

- Marine Pollution Bulletin
- Netherlands Journal of Sea Research

d) Reports from specialist websites searched

i) From Conservation Evidence discipline-wide literature database

All of the report series (and years) below have already been systematically 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 summarised.

• Amphibian Survival Alliance	1994-2012	Vol 9 - Vol 104
• British Trust for Ornithology	1981-2016	Report 1-687
• IUCN Invasive Species Specialist Group	1995-2013	Vol 1 - Vol 33
• Scottish Natural Heritage	2004-2015	Reports 1-945

ii) Update searches

Updates to reports already searched as part of the wider Conservation Evidence project were not undertaken for this synopsis.

No new report searches were undertaken for this synopsis due to time constraints.

e) Other literature searches

i) Conservation Evidence online database

The online database www.conservationevidence.com was searched for relevant publications that had already been summarised. If such summaries existed, they were extracted and added to this synopsis.

ii) Key word searches

Keyword searches were conducted on an additional three journals for the years 2000-2017, details of which are shown in Appendix 2a and Appendix 4.

iii) Systematic and non-systematic reviews

Where a systematic review was found for an intervention, it was summarised. However, each relevant study included in the systematic review was not summarised due to time constraints. Where a non-systematic review (or editorial, synthesis, preface, introduction etc.) was found for an intervention, the review itself was not summarised, unless the review also provided new/collective data. Relevant publications cited in these non-systematic reviews were not summarised at this stage.

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, where time permitted.

g) Search record database

A database was created of all relevant publications found during searches. Reasons for exclusion were recorded for all studies included during screening but were not summarised for the synopsis.

1.6.2. 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. The initial screening process is at the title and abstract level. If selected following this initial screening, a second one at the full-text level is undertaken, to validate whether the study indeed fits the Conservation Evidence inclusion criteria (described below).

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 the 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 and were tested again 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 has 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, results in gaps in the evidence. The Conservation Evidence literature database currently includes relevant papers from over 270 English language journals as well as over 150 non-English 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. Potential publication bias is not taken into account, and it is likely that additional biases will result from the evidence that is available, for example there are often 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 action that might be done to conserve biodiversity

1. Does this study measure the effect of an action that is or was under the control of humans, on wild taxa (including captives), habitats, or invasive/problem taxa? If yes, go to 3. If no, go to 2.
2. Does this study measure the effect of an action 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, exclude.
3. Could the action be put in place by a conservationist/decision maker to protect, manage, restore or reduce impacts of threats to wild taxa or habitats, or control or mitigate the impact of the invasive/problem taxon on wild taxa or habitats? If yes, include. If no, exclude.

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 actions that have a measured outcome on human behaviour only.

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

2. Study must test an action that could be put in place for conservation. This excludes assessing impacts of threats (actions which remove threats would be included). The test may involve comparisons between sites/factors not originally put in place or modified for invertebrate conservation, but which could be (e.g. modified fishing net vs unmodified fishing net, dredged sites vs sites where dredging stopped – where the net modification/dredge cessation 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 indicative of fulfilling our criteria, but you do not have sufficient information to judge whether the action was under human control, the action could be applied by a conservationist/decision maker or whether there are data quantifying the outcome, then include. If the article has no abstract, but the title is indicative that it might test a relevant intervention, then include. It is possible that some relevant publications are missed at this stage if the title is not deemed indicative by the author undertaking the search.

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 please scan the full article and then assign 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 include). 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/larvae/sperm production, mating success, birth rate, clutch size, , ‘overall recruitment’
- *Genetics*: genetic diversity, genetic suitability (e.g. adaptation to local conditions.)
- *Life history*: age/size at 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, physical habitat structure (e.g. rugosity, height, basal area)

Actions within the scope of Conservation Evidence include:

- Clear management actions: creation of artificial structures, planting submerged vegetation, controlling or eradicating invasive species, creating marine protected areas, creating or restoring habitats.
- International, national, or local policies: creation of marine protected area, bylaws, local voluntary restrictions.
- Reintroductions or management of wild species in captivity
- Actions that reduce human-wildlife conflict
- Actions that change human behaviour, resulting in an impact on wild taxa or habitats
- See https://www.conservationevidence.com/data/index for more examples of actions

Note on study types:
Include any literature reviews, systematic reviews, meta-analyses or short notes that review studies that fulfil these criteria.
Exclude theoretical modelling studies, as no action has been taken. However, studies that use models to analyse real-world data, or compare models to real-world situations are included (if they otherwise fulfil these criteria).

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

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

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. Action must be under human control: excludes impacts from climatic or other natural events.

2. Study must test an action that could be put in place for conservation: excludes studies with no action 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 include). It could be any behaviour that is likely to have an outcome on wild taxa and habitats (including mitigating the impact of invasive/problem taxon on wild taxa or habitats). Actions include, but are not limited to the following:

- *Change in adverse behaviours* (which directly threaten biodiversity): e.g. unsustainable or illegal fishing, 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. placement 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.

Actions which are particularly likely to induce a human behaviour change include, but are not limited to the following:

- *Enforcement*: Closed seasons, size limits, fishing/hunting gear restrictions, auditable/traceable reporting requirements, market inspections, increase number of rangers, patrols or frequency of patrols in, around or within protected areas, improve fencing/physical barriers, improve signage, improve equipment/technology used by guards, use of Unmanned Autonomous Vehicles/drones for rapid response, DNA analysis, GPS tracking.
- *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.
- *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 and regulations for conservation/protection:* Hard laws, soft laws, voluntary regulations.

We allocate studies to folders by their outcome. All studies under Criteria B go in the 'Human 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, file only in Human behaviour change).

Relevant subject

Studies relevant to the synopsis subject include those focussed on the conservation of wild, native marine subtidal benthic invertebrates and carried out in marine, brackish, and estuarine habitats.

Relevant types of intervention

An intervention has to be one that could be put in place by a marine conservationist, a community group, a marine protected area manager, or a policy maker, to protect, manage, restore or reduce the impacts of threats to wild native subtidal benthic invertebrate populations, or control or mitigate the impact of an invasive/problem taxon on subtidal benthic invertebrate populations. Alternatively, interventions may aim to change human behaviour (actual or intended), which is likely to protect, manage, restore or reduce threats to marine subtidal invertebrate populations. See inclusion criteria above for further details.

If the following two criteria were met, a combined intervention was created within the synopsis, rather than repeating evidence under all the separate interventions: a) there are five or more publications that use the same well-defined combination of interventions, with very clear descriptions 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.

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 comparing "treatment" sites where an intervention was undertaken or implemented, and "control" sites where not intervention took place but the threat occured.

Alternatively, a study could compare one specific intervention (or implementation method) against another. For example, this could be comparing the abundance of a species before and after the closure of an area to fishing activities, or the species selectivity or unwanted catch reduction of two different mesh sizes used in fishing gear.

Exceptions, which may not have one of the suitable comparators listed above, but will still be included, are for example, studies comparing with "pristine" or "reference" sites, or studies where no comparator is realistic (e.g: the effectiveness of restocking or captive breeding programmes, or of eradicating or controlling introduced species).

Relevant types of outcome

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

- Community response
    - Community composition
    - Richness/diversity
- Population response
    - Abundance: number, density, presence/absence, biomass, movement, age-structure, sex ratio
    - Reproductive success: egg/larvae production, mating success, hatching rate, egg/larvae quality/condition, overall recruitment, age/size at maturity
    - Survival: survival, mortality
    - Condition: growth, size, weight, condition factors (condition indices), biochemical ratios, stress, disease levels, or immune function
    - Unwanted catch abundance
- Behaviour:
    - Use by species of natural or artificial habitat, use of artificial structures or shelters
    - Species behaviour change: movement or migration patterns, changes in range,
    - Human behaviour change
- Other
    - Human-wildlife conflict: predatory or nuisance behaviour by species that could lead to retaliatory action by humans
    - Commercial catch abundance

Relevant types of study design

The table below lists the study designs included. The strongest evidence comes from studies using the following experimental design: randomized, replicated, controlled trials with paired-sites and before and after monitoring. For further information on study designs and their quality or strength, please see Christie *et al.* 2019.

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.
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 sediment type or surrounding seascape. 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 assessments of the evidence.
Systematic review	A systematic review follows an agreed set of methods for identifying studies and carrying out a formal 'meta-analysis'. It will weight or evaluate studies according to the strength of evidence they offer, based on the size of each study and the rigour of its design. All environmental systematic reviews are available at: www.environmentalevidence.org/index.htm

Study	Used if none of the above qualifiers apply. A “study” would, for example, look at the number of people that were engaged in an awareness raising project, or measure change over time in only one site and only after an intervention.

* 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 restored oyster reefs, compared to unrestored seabed plots (controlled) and natural, target oyster reefs (site comparison).

1.6.3. Study quality assessment & critical appraisal

We did not quantitatively assess the evidence from each publication or weigh it according to quality. However, to allow interpretation of the evidence, we clearly articulated the size and design of each reported study.

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 included during screening will be kept within the synopsis database.

1.6.4. Data extraction

Data on the performance/effect of the relevant intervention (e.g. mean species abundance inside or outside a closed area; reduction in unwanted catch after modifications of fishing nets) were extracted from, and summarised 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 scanned, and the total number of publications included following data extraction is presented in Appendix 3.

At the start of each month, authors swapped three summaries with another author to ensure that the correct type of data (e.g. correct comparator; relevant metrics…) has been extracted and that the summary followed the Conservation Evidence standard format.

1.6.5. Evidence synthesis

a) Summary protocol

Each publication usually had just one paragraph for each intervention it tested, describing the study in (usually) no more than 150 words using plain English. To help with some of the terminology specific to the marine environment, and for which Plain English equivalent do not exist, we provide a Glossary of terms (Appendix 1). Each summary used the following format:

A [TYPE OF STUDY] in [YEARS X-Y] in [HOW MANY SITES] in/of [HABITAT/SEABED TYPE] in [REGION, COUNTRY and WATER BODY] [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).

For example:

A replicated study in 2004 at four coral reef sites in the Singapore Strait (1) found that after being transplanted in the field aquarium-reared giant clams *Tridacna squamosa* had a high survival rate and grew over seven months. Of the 144 clams transplanted, 116 were recovered (80.6%), but survival rates differed amongst transplant sites (24–34 out of 36 transplanted clams per site). All recovered clams had increased in weight, shell length and shell height over the seven-month transplant, but rates differed amongst transplant sites (3.3–4.8 mm/month). In April 2004, a total of 144 aquarium-reared clams (eighteen-month old) were equally divided into 24 groups (6 clams/group) and transplanted into four sites (6 groups/site). Clams were released 50 cm above the seabed. Prior to transplant and after seven months, all clams were weighted, and their shell lengths and heights measured.

(1) Guest, J.R., Todd, P.A., Goh, E., Sivalonganathan, B.S., & Reddy, K.P. (2008) Can giant clam (*Tridacna squamosa*) populations be restored on Singapore's heavily impacted coral reefs? *Aquatic Conservation: Marine and Freshwater Ecosystems,* 18, 570–579.

A replicated, controlled study in 2003–2004 in the Varangerfjord, Norway (2) found that traps floated above the seabed caught fewer unwanted red king crabs *Paralithodes camtschaticus,* compared to standard groundfish traps. Red king crabs were only found in two of the 73 floated traps (2 and 3 crabs/trap), while all 77 standard traps caught crabs with an average catch of 21 crabs/trap. There was no difference in the number of marketable catches of the commercially targeted species, cod *Gadus morhua*, between the two trap designs. In August–September 2003 and 2004, sixteen lines of baited traps (100 x 150 x 120 cm) were deployed at 70–250 m depths. Two types of trap were used: a standard two-chamber groundfish trap and a floated version (approximately 70 cm above the seabed) of the same trap. Each line held five traps/design, placed alternatively. The traps were recovered after 24 hours, and catches sorted and counted.

(2) Furevik, D.M., Humborstad, O.B., Jørgensen, T. & Løkkeborg, S. (2008) Floated fish pot eliminates bycatch of red king crab and maintains target catch of cod. *Fisheries Research,* 92, 23–27.

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 a statistically significant difference or state that there was no difference if it was not significant. Table 1 above defines the terms used to describe the study designs.

c) Dealing with multiple interventions within a publication

When separate results are provided for the effects of each of the different interventions tested, separate summaries have been written under each intervention heading. However, when several interventions were carried out at the same time and only the combined effect reported, the results were described with a similar paragraph under all relevant interventions. The first sentence makes it clear that there was a combination of interventions carried out, i.e. '.........(REF) found that [x intervention], along with [y] and [z interventions] resulted in [describe effects]'. Within the results section we also added a sentence such as: 'It is not clear whether these effects were a direct result of [x], [y] or [z] interventions', or 'The study does not distinguish between the effects of [x], and other interventions carried out at the same time: [y] and [z].'

d) Dealing with multiple publications reporting the same results

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. if a study is published in an academic journal and in a report series, we would include the academic journal). 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

The taxonomy used in each summary paragraph was not updated but followed 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, which was written once all the identified literature had been summarised. These messages include information such as the number, design and location of included studies.
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]. *Here, locations include body of water and country, ordered based on chronological order of studies rather than alphabetically, i.e. Mediterranean Sea[1], Baltic Sea[2] not Baltic Sea[2], Mediterranean Sea[1]. The distribution of studies amongst specific habitat types or species groups may also be added here if relevant to the intervention.*

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)*

- **Commercial catch abundance (x studies):**
- **Human-wildlife conflicts (x studies):**
- **Biological production (x studies):**

1.6.6. Dissemination/communication of evidence synthesis

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

- This synopsis pdf, downloadable from www.conservationevidence.com, which contains the study summaries, key messages and background information on each intervention.
- The searchable database at www.conservationevidence.com which 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 [https://www.conservationevidence.com/content/page/79], which 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.

1.7 How you can help to change conservation practice

If you know of evidence relating to the conservation of subtidal benthic invertebrate communities that is not included in this synopsis, we invite you to contact us, via our website www.conservationevidence.com. You can submit a published study by clicking 'Submit additional evidence' on the right-hand side of an intervention page. If you have new, unpublished evidence, you can submit a paper to the *Conservation Evidence* journal. We particularly welcome papers submitted by conservation practitioners.

1.8 References

Amano T., González-Varo J.P. & Sutherland W.J. (2016) Languages are still a major barrier to global science. PLoS Biology, 14, e2000933.

Christie A.P., Amano T., Martin P.A., Shackelford G.E., Simmons B.I. & Sutherland W.J. (2019) Simple study designs in ecology produce inaccurate estimates of biodiversity responses. *Journal of Applied Ecology*, 00, 1– 13.

Gamfeldt L., Lefcheck J.S., Byrnes J.E., Cardinale B.J., Duffy J.E. & Griffin J.N. (2015) Marine biodiversity and ecosystem functioning: what's known and what's next? *Oikos*, 124, 252–265.

Lotze H.K., Guest H., O'Leary J., Tuda A. & Wallace D. (2018) Public perceptions of marine threats and protection from around the world. *Ocean & Coastal Management*, 152, 14–22.

Sutherland W.J., Taylor N.G., MacFarlane D., Amano T., Christie A.P., Dicks L.V., Lemasson A.J., Littlewood N.A., Martin P.A., Ockendon N., Petrovan S.O., Robertson R.J., Rocha R., Shackelford G.E., Smith R.K., Tyler E.H.M. & Wordley C.F.R. (2019) Building a tool to overcome barriers in the research-implementation space: The conservation evidence database. *Biological Conservation*, 238, 108–199.

2. Threat: Residential and commercial development

Background

The greatest threats from development tend to be destruction of habitat, pollution, and impacts from activities related to energy production and transportation. Interventions in response to these threats are described in other chapters and therefore will not be repeated here, please refer to the following chapters: “Habitat protection”, “Habitat restoration and creation”, “Threat: Pollution”, “Threat: Energy production and mining” and “Threat: Transportation and service corridors”.

3. Threat: Aquaculture and Agriculture

Background

Marine aquaculture is the farming of fish, crustaceans, molluscs, algae and other organisms under controlled conditions in the marine environment. Aquaculture can impact subtidal seabed communities through direct damage to the seabed from the construction of aquaculture facilities, or indirectly due to accumulated pollution from biological waste, food or chemicals used in aquaculture systems (Anderson 2005), or through the spread of non-native, problematic or invasive species (Gallardi 2014). The impacts that aquaculture has on the seabed in terms of physical damage and shading from infrastructures, and pollution from overfeeding and biological wastes, tends to be limited to the direct locality of the operations (Johannessen *et al.* 1994).

Nutrient-rich and pesticide-rich run-offs from land agriculture reach the marine environment through rivers, and negatively impact coastal areas due to the increase in nutrients such as nitrogen and phosphorous (Falace *et al.* 2018; Gabric & Bell 1993). These increases in nutrients often lead to diminished water quality and eutrophication events including hypoxia or anoxia, creating "*dead zones*" (Breitburg *et al.* 2018).

Much of the conservation effort related to threats from aquaculture and agriculture has been directed at reducing the impacts of pollution and impoverished water quality, as well as reducing the threat from non-native and invasive species. Interventions related to these threats are described in "Threat: Pollution" and "Threat: Non-native, invasive and problematic species" and are not repeated here. Other interventions related to reducing or mitigating the impacts from aquaculture or aiming to promote sustainable practices, but where effects on benthic subtidal invertebrates were not necessarily tested or reported, are summarised in the Sustainable Aquaculture synopsis.

Anderson R. (2005). Environmental effects of marine finfish aquaculture (Vol. 5). Springer Science & Business Media.

Breitburg D., Levin L.A., Oschlies A., Grégoire M., Chavez F.P., Conley D.J., Garçon V., Gilbert D., Gutiérrez D., Isensee K., Jacinto G.S., Limburg K.E., Montes I., Naqvi S.W.A., Pitcher G.C., Rabalais N.N., Roman M.R., Rose K.A., Seibel B.A., Telszewski M., Yasuhara M. & Zhang J. (2018) Declining oxygen in the global ocean and coastal waters. *Science*, 359.

Falace A., Tamburello L., Guarnieri G., Kaleb, S., Papa L. & Fraschetti S. (2018) Effects of a glyphosate-based herbicide on Fucus virsoides (Fucales, Ochrophyta) photosynthetic efficiency. *Environmental Pollution*, 243, 912–918.

Gabric A.J. & Bell P.R.F. (1993) Review of the effects of non-point nutrient loading on coastal ecosystems. *Marine and Freshwater Research*, 44, 261–283.

Gallardi D. (2014) Effects of bivalve aquaculture on the environment and their possible mitigation: a review. *Fisheries and Aquaculture Journal*, 5, 1.

Johannessen P., Botnen H. & Tvedten Ø.F. (1994) Macrobenthos: before, during and after a fish farm. *Aquaculture Research*, 25, 55–66.

4. Threat: Energy production and mining

Background

Energy production (renewable and non-renewable), mining (for minerals), quarrying, and aggregate extraction, can have significant impacts on subtidal benthic invertebrates through the modification, destruction and pollution of seabed habitats during construction, routine activities, and decommissioning (Boehlert & Gill 2010; Newell *et al*. 2004). Additional threat arises from the spread of non-native and invasive species colonising offshore infrastructures associated with these activities. Interventions related to recreating or re-establishing natural habitats following activities or related to repurposing infrastructure as artificial habitats (Langhamer 2012) are described in the chapter “Habitat restoration and creation”. Interventions related to pollution emanating from energy production and mining, including noise generation, are described in “Threat: Pollution”. Interventions related to the introduction and spread of non-native, invasive or problematic species due to the “*stepping stones*” effects associated with installations and anthropogenic structures are described in “Threat: Non-native, invasive and problematic species”.

Interventions in response to other threats caused by energy production and mining are covered below. Note that at the time of writing, deep-sea mining for minerals is not yet taking place, and as such related threats and conservation actions are not mentioned here. Note also that pre-emptive actions aiming to prevent the occurrence of a threat at a location (e.g. “locate cables or infrastructure away from sensitive areas”) are not described here, as robustly testing for their effect would not be feasible. However, pre-emptive management actions that can be undertaken at the planning stage before an activity takes place and aiming to reduce the likelihood or level of a threat (e.g. “Limit the thickness of drill cuttings”) are included in this chapter.

Boehlert G., & Gill A. (2010) Environmental and ecological effects of ocean renewable energy development: A current synthesis. *Oceanography*, 23, 68–81.

Langhamer O. (2012) Artificial reef effect in relation to offshore renewable energy conversion: State of the art. *The Scientific World Journal*.

Newell R.C., Seiderer L.J., Simpson N.M., & Robinson J.E. (2004) Impacts of marine aggregate dredging on benthic macrofauna off the south coast of the United Kingdom. *Journal of Coastal Research*, 20, 115–125.

General

4.1. Set limits for change in sediment particle size during rock dumping

- We found no studies that evaluated the effects of setting limits for change in sediment particle size during rock dump on subtidal benthic invertebrate 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

Rock dump involves placing rock material on the seabed or surrounding infrastructure to stabilise underwater structures from offshore industries, such as oil and gas platforms or windfarms, to protect pipelines, and remove the risk of snagging by fishing vessels operating trawl nets (Visser & van der Meer 2008). Rock dump can impact subtidal benthic invertebrates through loss of natural sediment and changes in habitat characteristics, such as particle size. Setting limits for changes in particle size during rock dump may reduce the level of threat and retain suitable sediment and habitat properties, thereby potentially reducing risk to subtidal benthic invertebrates. Evidence related to other means of stabilizing or burying offshore infrastructures and pipelines are summarised under "Threat: Energy production and mining – Bury pipelines instead of surface laying and rock dumping" and "Threat: Transportation and service corridors – Bury cables and pipelines in the seabed rather than laying them on the seabed". Other evidence for interventions related to rock dumping are summarised in "Habitat restoration and creation – Modify rock dump to make it more similar to natural substrate".

Visser, R. & van der Meer, J. (2008) Immediate displacement of the seabed during Subsea Rock Installation (SRI). *Terra et Aqua*, 110.

4.2. Bury pipelines instead of surface laying and rock dumping

- We found no studies that evaluated the effects of burying pipelines instead of surface laying and rock dumping on subtidal benthic invertebrate 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

Rock dump involves placing rock material on the seabed or surrounding infrastructure to stabilise underwater structures from offshore industries, such as oil and gas platforms or windfarms, to protect pipelines, and remove the risk of snagging by fishing vessels operating trawl nets (Visser & van der Meer 2008). Rock dump can impact subtidal benthic invertebrates through loss of natural sediment and changes in habitat characteristics, such as particle size. Burying pipelines removes the need for dumping rock over them as a protection means, therefore potentially reducing the level of associated threats to subtidal benthic invertebrates (De Groot 1982 ; Morrow & Larking 2007). Evidence for other interventions related to rock dumping are summarised under "Threat: Energy production and mining – Use stabilisation material that can be more easily recovered at decommissioning stage" and "Limit the amount of stabilisation material used". Evidence for other interventions related to pipelines and subsea cables are summarised under "Habitat restoration and creation – Cover subsea cables with artificial reefs" and "Cover subsea cables with materials that encourage the accumulation of natural sediments".

De Groot S.J. (1982) The impact of laying and maintenance of offshore pipelines on the marine environment and the North Sea fisheries. *Ocean Management*, 8, 1–27.
Morrow D.R., & Larkin P.D. (2007) The challenges of pipeline burial. In *The Seventeenth International Offshore and Polar Engineering Conference*. International Society of Offshore and Polar Engineers.

4.3. Limit the amount of stabilisation material used

- We found no studies that evaluated the effects of limiting the amount of stabilisation material used on subtidal benthic invertebrate 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

Stabilisation material, such as concrete mattresses and rocks (rock dump), can be used to keep pipelines and infrastructure (from offshore industry structures, such as oil and gas, renewable energy, and mining or aggregate extraction) in place and may also help to protect the infrastructure (for instance from fishing gears) by covering them. This process and material can impact subtidal benthic invertebrates through physical disturbances, loss of natural sediment, and changes in habitat characteristics. Limiting the amount of stabilisation material used can reduce the threat to subtidal benthic invertebrates by reducing the amount of physical damage and habitat loss. Evidence for other interventions related to the use of stabilisation material are summarised under "Threat: Energy production and mining – Use stabilisation material that can be more easily recovered at decommissioning stage".

4.4. Use stabilisation material that can be more easily recovered at decommissioning stage

- We found no studies that evaluated the effects of using stabilisation material that can be more easily recovered at decommissioning stage on subtidal benthic invertebrate 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

Stabilisation material, such as concrete mattresses and rocks (rock dump), can be used to keep pipelines and infrastructure (from offshore industry structures, such as oil and gas, renewable energy, and mining or aggregate extraction) in place and may also help to protect the infrastructure (for instance from fishing gears) by covering them. Stabilisation material may need to be recovered following decommissioning, and their retrieval might cause additional physical disturbances to the seabed and to subtidal benthic invertebrates. Stabilisation materials designed to be easily recovered at the decommissioning stage can help reduce the level of habitat disturbance associated with this process. Evidence for interventions related to the decommissioning of offshore infrastructures and pipelines are summarised under "Threat: Energy production and mining – Remove pipelines and infrastructure following decommissioning" and "Leave pipelines and infrastructure in place following decommissioning".

4.5. Leave pipelines and infrastructure in place following decommissioning

• We found no studies that evaluated the effects of leaving pipelines and infrastructure in place following decommissioning on subtidal benthic invertebrate 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 and infrastructure from offshore industry structures, such as oil and gas, renewable energy, and mining can impact subtidal benthic invertebrates through physical damage and loss of habitat when the infrastructure is constructed, but also when they are removed following decommissioning. Leaving structures in place may involve the least environmental and physical disturbances (Brigitte *et al*. 2018; Chandler *et al*. 2017). In addition, it could potentially benefit subtidal benthic invertebrates by providing habitat and shelter (Ponti *et al.* 2002; see also related interventions: "Habitat restoration and creation – Repurpose obsolete offshore structures to act as artificial reefs", "Cover subsea cables with artificial reefs", and "Cover subsea cables with materials that encourage the accumulation of natural sediments").

Brigitte S., Fowler A.M., Macreadie P.I., Palandro D.A., Aziz A.C. & Booth D.J. (2018) Decommissioning of offshore oil and gas structures–Environmental opportunities and challenges. *Science of the Total Environment*, 658, 973–981.

Chandler J., White D., Techera E. J., Gourvenec S. & Draper S. (2017) Engineering and legal considerations for decommissioning of offshore oil and gas infrastructure in Australia. *Ocean Engineering*, 131, 338–347.

4.6. Remove pipelines and infrastructure following decommissioning

• We found no studies that evaluated the effects of removing pipelines and infrastructure in place following decommissioning on subtidal benthic invertebrate 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 and infrastructure from offshore industry structures, such as oil and gas, renewable energy, and mining can impact subtidal benthic invertebrates through physical damage and loss of habitat when the infrastructure is constructed, but also following decommissioning through indirect environmental impacts, such as chemical leaching and corrosion of structures (Picken *et al.* 1997). Removing structures is one decommissioning option to remove the threat from the marine environment (Brigitte *et al*. 2018; Chandler *et al.* 2017). In the OSPAR maritime area for instance, the at-sea disposal or leaving in place of disused offshore installations is prohibited (OSPAR 1998).

Brigitte S., Fowler A.M., Macreadie P.I., Palandro D.A., Aziz A.C. & Booth D.J. (2018) Decommissioning of offshore oil and gas structures–Environmental opportunities and challenges. *Science of the Total Environment*, 658, 973–981.

Chandler J., White D., Techera E. J., Gourvenec S. & Draper S. (2017) Engineering and legal considerations for decommissioning of offshore oil and gas infrastructure in Australia. *Ocean Engineering*, 131, 338–347.

OSPAR Decision 98/3 (1998) On the Disposal of Disused Offshore Installations. In: Ministerial Meeting of the OSPAR Commission. OSPAR Convention for the protection of the marine environment of the North-East Atlantic, London, UK.
Picken G., Curtis T. & Elliott A. (1997) An estimate of the cumulative environmental effects of the disposal in the deep sea of bulky wastes from the offshore oil and gas industry. In: *Offshore Europe*. Society of Petroleum Engineers.

Oil and gas drilling

4.7. Cease or prohibit oil and gas drilling

- We found no studies that evaluated the effects of ceasing or prohibiting oil and gas drilling on subtidal benthic invertebrate 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

Routine oil and gas drilling activities can impact subtidal invertebrate seabed communities due to smothering and burial from drill cuttings and drill fluids, , pollution from the use of chemicals and additives, physical damage or loss of suitable natural sediment (Cordes *et al*. 2016). Ceasing on-going oil and gas drilling, for instance following protective legislation or the non-renewal of permit, can stop the threat and potentially allow for the community to recover over time.

However, it should be kept in mind that prohibition in one place may simply lead to displacement, which may impact the same or different communities in other locations.

Cordes E.E., Jones D.O., Schlacher T.A., Amon D.J., Bernardino A.F., Brooke S., Carney R., DeLeo D.M., Dunlop K.M., Escobar-Briones E.G. & Gates A.R. (2016) Environmental impacts of the deep-water oil and gas industry: a review to guide management strategies. *Frontiers in Environmental Science*, 4, 58.

4.8. Cease or prohibit the deposit of drill cuttings on the seabed

- We found no studies that evaluated the effects of ceasing or prohibiting the deposit of drill cuttings on the seabed on subtidal benthic invertebrate 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

Routine oil and gas drilling activities can impact subtidal invertebrate seabed communities due to the production of drill cuttings. Drill cuttings consist of the fragments of rock that are removed as each oil or gas well is drilled. The drill cuttings are usually discharged onto the seafloor in the vicinity of the platforms to form a cuttings pile, but are often contaminated with drilling fluids, oil and chemical additives which can leach and pollute the sediments. Drill cuttings can also smother and bury organisms under their weight (Henry *et al.* 2017). Ceasing the deposit of drill cuttings, for instance following protective legislation or changes in activity management, can stop the threat and potentially allow for the community to recover over time. Evidence related to alternative means of disposing of drill cuttings are summarised under “Threat: Energy production and mining - Dispose of drill cuttings on land rather than on the seabed” and “Bury drill cuttings in the seabed rather than leaving them on the seabed surface”.

Henry L.A., Harries D., Kingston P. & Roberts J.M. (2017) Historic scale and persistence of drill cuttings impacts on North Sea benthos. *Marine Environmental Research,* 129, 219–228.

4.9. Dispose of drill cuttings on land rather than on the seabed

- We found no studies that evaluated the effects of disposing of drill cuttings on land rather than on the seabed on subtidal benthic invertebrate 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

Routine oil and gas drilling activities can impact subtidal invertebrate seabed communities due to the production of drill cuttings. Drill cuttings consist of the fragments of rock that are removed as each oil or gas well is drilled. The drill cuttings are usually discharged onto the seafloor in the vicinity of the platforms to form a cuttings pile, but are often contaminated with drilling fluids, oil and chemical additives which can leach and pollute the sediments. Drill cuttings can also smother and bury organisms under their weight (Henry *et al*. 2017). Disposing of drill cuttings on land rather than on the seabed (Melton *et al.* 2000; 2004), for instance following protective legislation or changes in activity management, can potentially stop the threat and allow for the community to recover over time. Evidence related to alternative means of disposing drill cuttings are summarised under “Threat: Energy production and mining - Bury drill cuttings in the seabed rather than leaving them on the seabed surface”, and those related to stopping their disposal under “Cease or prohibit the deposit of drill cuttings on the seabed”.

Henry L.A., Harries D., Kingston P. & Roberts J.M. (2017) Historic scale and persistence of drill cuttings impacts on North Sea benthos. *Marine Environmental Research,* 129, 219–228.

Melton H.R., Smith J.P., Mairs H.L., Bernier R.F., Garland E., Glickman A.H., Jones F.V., Ray J.P., Thomas D. & Campbell J.A. (2004) *Environmental aspects of the use and disposal of non-aqueous drilling fluids associated with offshore oil & gas operations*. In: SPE International Conference on Health, Safety, and Environment in Oil and Gas Exploration and Production. Society of Petroleum Engineers.

Melton H.R., Smith J.P., Martin C.R., Nedwed T.J., Mairs H.L. & Raught D.L. (2000) *Offshore discharge of drilling fluids and cuttings–a scientific perspective on public policy*. In Rio Oil and Gas Conference. Rio de Janeiro, Brazil.

4.10. Remove drill cuttings after decommissioning

- We found no studies that evaluated the effects of removing drill cuttings after decommissioning on subtidal benthic invertebrate 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

Routine oil and gas drilling activities can impact subtidal invertebrate seabed communities due to the production of drill cuttings. Drill cuttings consist of the fragments of rock that are removed as each oil or gas well is drilled. The drill cuttings are usually discharged onto the seafloor in the vicinity of the platforms to form a cuttings pile, but are often contaminated with drilling fluids, oil and chemical additives which can leach and pollute the sediments. Drill cuttings can also smother and bury organisms under their weight (Henry *et al.* 2017). Removing discarded drill cuttings from the seabed following decommissioning (Melton *et al.* 2000; 2004), instead of leaving them on the seabed, can stop the threat and potentially allow for the community to recover over time. Evidence for a related intervention, relating to mine tailings remediation following decommissioning, are summarised under "Threat: Energy production and mining - Leave mine tailings in place following cessation of activities (submarine tailing disposal operations)".

Henry L.A., Harries D., Kingston P. & Roberts J.M. (2017) Historic scale and persistence of drill cuttings impacts on North Sea benthos. *Marine Environmental Research,* 129, 219–228.

Melton H.R., Smith J.P., Mairs H.L., Bernier R.F., Garland E., Glickman A.H., Jones F.V., Ray J.P., Thomas D. & Campbell J.A. (2004) *Environmental aspects of the use and disposal of non-aqueous drilling fluids associated with offshore oil & gas operations*. In: SPE International Conference on Health, Safety, and Environment in Oil and Gas Exploration and Production. Society of Petroleum Engineers.

Melton H.R., Smith J.P., Martin C.R., Nedwed T.J., Mairs H.L. & Raught D.L. (2000) *Offshore discharge of drilling fluids and cuttings–a scientific perspective on public policy*. In Rio Oil and Gas Conference. Rio de Janeiro, Brazil.

4.11. Limit the thickness of drill cuttings

- We found no studies that evaluated the effects of limiting the thickness of drill cuttings on subtidal benthic invertebrate 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

Routine oil and gas drilling activities can impact subtidal invertebrate seabed communities due to the production of drill cuttings. Drill cuttings consist of the fragments of rock that are removed as each oil or gas well is drilled. The drill cuttings are usually discharged onto the seafloor in the vicinity of the platforms to form a cuttings pile, but are often contaminated with drilling fluids, oil and chemical additives which can leach and pollute the sediments. Drill cuttings can also smother and bury organisms under their weight (Henry *et al*. 2017). Limiting the thickness of drill cuttings can potentially reduce the level of threat to subtidal benthic invertebrates (Trannum *et al*. 2011).

Henry L.A., Harries D., Kingston P. & Roberts J.M. (2017) Historic scale and persistence of drill cuttings impacts on North Sea benthos. *Marine Environmental Research,* 129, 219–228.

Trannum H.C., Setvik Å., Norling K. & Nilsson H.C. (2011) Rapid macrofaunal colonization of water-based drill cuttings on different sediments. *Marine Pollution Bulletin*, 62, 2145–2156.

4.12. Bury drill cuttings in the seabed rather than leaving them on the seabed surface

- We found no studies that evaluated the effects of burying drill cuttings in the seabed rather than leaving them on the seabed surface on subtidal benthic invertebrate 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

Routine oil and gas drilling activities can impact subtidal invertebrate seabed communities due to the production of drill cuttings. Drill cuttings consist of the fragments of rock that are removed as each oil or gas well is drilled. The drill cuttings are usually discharged onto the seafloor in the vicinity of the platforms to form a cuttings pile, but are often contaminated with drilling fluids, oil and chemical additives which can leach and pollute the sediments. Drill cuttings can also smother and bury organisms under their weight (Henry *et al*. 2017). Burying drill cuttings deep inside the sediment, a process, referred to as "cuttings re-injection" or "drill cuttings sub-surface injection" (Gumarov *et al.* 2014; Melton *et al.* 2000, 2004; Shadizadeh *et al.* 2011), rather than depositing them on the surface of the sediment, can potentially reduce the level of threat occurring for surface subtidal benthic invertebrates and those living inside the sediments at shallow depths. Evidence related to alternative means of disposing drill cuttings are summarised under "Threat: Energy production and mining – Dispose of drill cuttings on land rather than on the seabed", and those related to stopping their disposal under "Cease or prohibit the deposit of drill cuttings on the seabed".

Gumarov S.M., Shokanov T.A., Simmons S., Anokhin V.V., Benelkadi S. & Ji L. (2014) *Drill cuttings re-injection well design and completion: Best practices and lessons learned*. Society of Petroleum Engineers.

Henry L.A., Harries D., Kingston P. & Roberts J.M. (2017) Historic scale and persistence of drill cuttings impacts on North Sea benthos. *Marine Environmental Research,* 129, 219–228.

Melton H.R., Smith J.P., Mairs H.L., Bernier R.F., Garland E., Glickman A.H., Jones F.V., Ray J.P., Thomas D. & Campbell J.A. (2004) *Environmental aspects of the use and disposal of non-aqueous drilling fluids associated with offshore oil & gas operations*. In: SPE International Conference on Health, Safety, and Environment in Oil and Gas Exploration and Production. Society of Petroleum Engineers.

Melton H.R., Smith J.P., Martin C.R., Nedwed T.J., Mairs H.L. & Raught D.L. (2000) *Offshore discharge of drilling fluids and cuttings–a scientific perspective on public policy*. In Rio Oil and Gas Conference. Rio de Janeiro, Brazil.

Shadizadeh S.R., Majidaie S. & Zoveidavianpoor M. (2011) Investigation of drill cuttings reinjection: Environmental management in Iranian Ahwaz Oilfield. *Petroleum Science and Technology*, 29, 1093–1103.

4.13. Use water-based muds instead of oil-based muds (drilling fluids) in the drilling process

- We found no studies that evaluated the effects of using water-based muds instead of oil-based muds (drilling fluids) in the drilling process on subtidal benthic invertebrate 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

Fluids used in the drilling process, also called "muds" are often contaminated with oil and chemical additives causing pollution in the area (Henry *et al.* 2017). Traditionally-used oil-based muds and the discharge in water or sediment of cuttings contaminated with them are now prohibited in the OSPAR region. Using water-based muds where applicable as an alternative could potentially help to significantly reduce the pollution related environmental risks to subtidal benthic invertebrates associated with drill cuttings (OSPAR 2000; Patel *et al*. 2007). Additional evidence related to drilling fluids are summarised under "Threat: Energy production and mining – Recycle or repurpose fluids used in the drilling process".

Henry L.A., Harries D., Kingston P. & Roberts J.M. (2017) Historic scale and persistence of drill cuttings impacts on North Sea benthos. *Marine Environmental Research,* 129, 219–228.
Ospar Commission. (2000) OSPAR Decision 2000/3 on the use of rganic-Phase Drilling Fluids (OPF) and the discharge of OPF-contaminated cuttings. OSPAR, Copenhagen, Denmark.
Patel A., Stamatakis S., Young S. & Friedheim J. (2007) *Advances in inhibitive water-based drilling fluids—can they replace oil-based muds?* In: International Symposium on Oilfield Chemistry. Society of Petroleum Engineers.

4.14.Recycle or repurpose fluids used in the drilling process

• We found no studies that evaluated the effects of recycling or repurposing fluids used in the drilling process on subtidal benthic invertebrate 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

Fluids used in the drilling process, also called "muds" are often contaminated with oil and chemical additives causing pollution in the area (Henry *et al.* 2017). These fluids could be reused or recycled, potentially reducing the cumulative risk to subtidal benthic invertebrates through reducing a source of pollution (Loan *et al*. 2018). Additional evidence related to drilling fluids are summarised under "Threat: Energy production and mining – Use water-based muds instead of oil-based muds (drilling fluids) in the drilling process".

Henry L.A., Harries D., Kingston P. & Roberts J.M. (2017) Historic scale and persistence of drill cuttings impacts on North Sea benthos. *Marine Environmental Research,* 129, 219–228.
Loan M.E., Herron M., Akkurt R., Pomerantz A.E. & Schlumberger Technology Corp. (2018) Oil-based mud drill cutting cleaning for infrared spectroscopy. U.S. Patent Application 15/410,045.

Mining, quarrying, and aggregate extraction

4.15.Cease or prohibit aggregate extraction

• **Seven studies** examined the effects of ceasing or prohibiting aggregate extraction on subtidal benthic invertebrate populations. One study was in the English Channel[1] (France), one in the Mediterranean Sea[2] (Italy), one a global study[4], and four in the North Sea[3,5,6,7] (UK, Belgium).

COMMUNITY RESPONSE (6 STUDIES)

- **Overall community composition (4 studies):** One global systematic review[4] found that it took nine months to several decades for overall invertebrate community composition to recover after ceasing aggregate extraction. One before-and-after, site comparison study in the Mediterranean Sea[2] and one of two site comparison studies in the North Sea[6,7] found that after ceasing aggregate extraction overall invertebrate community composition became more similar to pre-extraction and/or natural site communities.
- **Overall richness/diversity (5 studies):** Two before-and-after, site comparison studies in the English Channel[1] and the Mediterranean Sea[2] and one of two site comparison studies in the North Sea[6,7] found that after ceasing aggregate extraction, overall invertebrate species richness and/or diversity became more similar to that of pre-extraction and/or natural sites[7]. The other site comparison found that species richness did not change over time and remained different to that of natural sites[6]. One replicated, site comparison study in the North Sea[5] found that 21 months after ceasing aggregate extractiom, invertebrate species richness was similar to that of natural sites.
- **Worm community composition (1 study):** One before-and-after study in the North Sea[3] found that after ceasing aggregate extraction, nematode worm community composition remained different to the pre-extraction community.
- **Worm richness/diversity (1 study):** One before-and-after study in the North Sea[3] found that after ceasing aggregate extraction, nematode worm species richness remained different to pre-extraction richness.

POPULATION RESPONSE (6 STUDIES)

- **Overall abundance (5 studies):** Two before-and-after, site comparison studies in the English Channel[1] and the Mediterranean Sea[2] and one of two site comparison studies in the North Sea[6,7] found that after ceasing aggregate extraction overall invertebrate abundance and/or biomass became more similar to that of pre-extraction and/or natural sites[7]. The other site comparison found that abundance and biomass did not change over time and remained different to that of natural sites[6]. One replicated, site comparison study in the North Sea[5] found that 21 months after ceasing aggregate extractiom, invertebrate abundance was similar to that of natural sites.
- **Worm abundance (1 study):** One before-and-after study in the North Sea[3] found that after ceasing aggregate extraction, nematode worm abundance remained different to pre-extraction abundance.

Background

'Aggregates' is the collective term for sand, gravel and crushed rock. They are used as raw materials for the construction industry as well as for beach replenishment schemes. Aggregate extraction can impact subtidal benthic invertebrates through physical removal, loss and alteration of seabed and habitat, and direct physical damage from the machinery used (De Groot 1996). Aggregate extraction could be ceased (for instance following end of licence or voluntary cessation of activities) or prohibited (through legislation) in an area, and the site and its invertebrates left to naturally recover over time (Desprez 2000). Evidence for other interventions related to aggregate extraction is summarised under "Threat: Energy production and mining – Mining and quarrying", and in "Habitat restoration and creation – Refill disused borrow pits" and "Landscape or artificially enhance the seabed".

De Groot S.J. (1996) The physical impact of marine aggregate extraction in the North Sea. *ICES Journal of Marine Science*, 53, 1051–1053.

Despres M. (2000) Physical and biological impact of marine aggregate extraction along the French coast of the Eastern English Channel: short-and long-term post-dredging restoration. *ICES Journal of Marine Science,* 57, 1428–1438.

A before-and-after, site comparison study in 1994–1997 of eight sites in one area of sandy and gravelly seabed in the English Channel, off the coast of France (1) found that 16–28 months after ceasing aggregate extraction, invertebrate species richness, abundance, and biomass appeared to have increased, and had become more similar to that of adjacent natural sites where extraction did not occur. Data were not statistically tested. After cessation, species richness at extraction sites increased and appeared to be more similar to natural sites (before: 37% that of natural sites; after 16 months onwards: >100%). Increases were also observed for biomass (before: 17%; after 16 months: 35%; after 28 months: 75% that of natural sites) and abundance (before: 14%; after 16 months: 56%; after 28 months: 57% that of natural sites). Aggregate extraction took place between 1980 and 1994. In 1994 (prior to cessation), 1996 (after 16 months) and 1997 (after 28 months), invertebrate communities were surveyed at five extracted sites and three natural sites (1 km outside the extraction area). Samples were collected using a sediment grab (0.1 m^2; three samples/site/year) and invertebrates (>1 mm) identified, counted, and dry-weighed.

A before-and-after, site comparison study in 2001–2004 of 10 sites in one area of sandy seabed in the northern Mediterranean Sea, Italy (2) found that from 18 months after ceasing aggregate extraction, invertebrate community composition, species richness, abundance and diversity were more similar to that of pre-extraction, and adjacent sites where extraction did not occur. Similarity in community composition with that of pre-extraction increased over time following cessation of extraction, from 9% similarity after one month, to 41–48% after 18–30 months. In addition, community composition became statistically similar to that of adjacent unextracted natural sites over time (59–66% similarity). Invertebrate species richness at extraction sites after 30 months (60–78) was similar to before extraction (54–72), and to unextracted sites (72–79). This was also true for invertebrate abundance (after: 2,300–2,500; before: 1,400–2,400; unextracted: 1,800–2,900 individual/m^2), and diversity (as a diversity index) (these data were not statistically tested). Three sites (40–42 m depth) were dredged for aggregate extraction in April–May 2002. These and seven adjacent unextracted sites were surveyed once before (March 2001) and once during (April 2002) dredging, and six times following cessation (after one, six, 12, 18, 24 and 30 months). Three sediment samples/site/survey were collected using a grab (24 dm^3) and pooled. Invertebrates (>0.5 mm) were identified and counted.

A before-and-after study in 1978–2004 in one area of soft seabed of the Kwintebank, North Sea, Belgium (3) found that following prohibition of aggregate extraction, nematode worm community composition changed but remained different to that of pre-extraction after 8–12 months, and worm abundance and diversity did not change. Community composition after cessation was different to that of during intense extraction, but also to that of before intense extraction began (presented as graphical analyses). Worm abundance and diversity were similar before and 8–12 months after extraction stopped and ranged between 84 and 228/10 cm^2 (abundances for each time period not presented; diversity reported as 10 different indices). In February 2003, extraction was prohibited where aggregate extraction had occurred since 1976. Two to three stations were sampled in 1978 (prior extraction intensification), in 1997 and 2001 (during intense extraction), and in October 2003 and February 2004 (eight and 12 months after extraction stopped, respectively). Sediment samples were collected using a 10 cm^2 core, and nematode worms (between 38 μm and 1 mm) identified and counted.

A systematic review of 22 case studies reported between 1977 and 2007 of marine aggregate extraction sites across the world (4) found that, after extraction stopped, invertebrate communities took between nine months (Bristol Channel, UK) and several decades (Thames, UK) to 'recover' (terminology not explained) and become similar to communities occurring at non-impacted sites or prior to-extraction. Invertebrate community recovery time varied with seabed type and current strength (data not statistically tested). The shortest average 'recovery' time (4.5 years) was recorded for shallow mixed sediment plains with moderate currents (1.8–4 Nm^2). The longest average 'recovery' time (10.8 years) was recorded for shallow coarse sediment plains with weak currents (0–1.8 Nm^2). Case studies were identified by using set search terms, and included peer-reviewed publications (n=18), technical reports (n=2), unpublished data (n=1) and personal communication (n=1). Aggregate extraction sites were categorised by seabed type and current strength. Invertebrate 'recovery' times were extracted from community composition, abundance, biomass and diversity data for each site (data not presented).

A site comparison study in 2004 of seven sites of sandy seabed in the southern North Sea, off the coast of Belgium (5) found that 21 months after ceasing aggregate extraction at sites, invertebrate species richness and abundance were similar to that of nearby natural sites where extraction did not occur. Extracted sites had similar number of invertebrate species (16–18/site) compared to natural sites (12–17/site) after 21 months, and in similar abundance (extracted: 700–990 individuals/m^2; natural: 480–860 individuals/m^2). In February 2003, aggregate extraction ceased in the Kwinte Bank licence area. In November 2004, invertebrates were surveyed at three extracted sites and at four natural sites in the nearby Middelkerke Bank. Five samples/sites were collected using a sediment grab, and invertebrates (>1 mm) were identified and counted.

A site comparison study in 2001–2004 of four sites of sandy seabed in the southern North Sea, UK (6) found that ceasing aggregate extraction did not lead to changes in invertebrate community composition, or increases in species richness, biomass or abundance, after five years, which all remained different to that of two natural sites where extraction did not occur. Invertebrate community composition did not change from one year to another at any of the sites. After five years, community composition at the extracted sites was only 32.5% similar to that of the natural sites. In addition, average invertebrate species richness, biomass and abundance did not change from one year to another at any of the sites, and was consistently lower at the extracted sites (richness: 12–20 species/sample in 2001, 12–15 in 2004; biomass: 0.03 g/sample in 2001, 0.12 in 2004; abundance: 20–42 individuals/samples in both 2001 and 2004) compared to the natural sites (richness: 55 in 2001, 45 in 2004; biomass: 0.58 in 2001, 0.32 in 2004; abundance: 141 in 2001, 563 in 2004). In 1999, aggregate extraction ceased in a licence area. Between 2001–2004, invertebrates were surveyed yearly at two extracted sites within the licence area and two natural sites 1–15 km outside. Ten samples/survey/extracted sites and five/survey/natural sites were collected using a sediment grab (0.1 m^2). Invertebrates (>0.5 mm) were identified, counted and dry-weighed.

A site comparison study in 2001–2011 of three sites in one area of sandy seabed off the southeast coast of England, North Sea, UK (7) found that, 15 years after ceasing aggregate extraction and letting the seabed recover naturally, invertebrate community composition, species richness, abundance and biomass were similar to that of adjacent sites where extraction did not occur. Although still different after five years, invertebrate community composition at the extraction site became more similar to that of the non-extraction sites over time and was undistinguishable after 15 years (data presented as graphical analyses and statistical model results). After 15 years, extraction and non-extraction sites had similar invertebrate species richness (55 vs 62), abundance (171 vs 183 individuals/0.1 m^2), and biomass (0.6 vs 0.7 g/0.1 m^2). In 2011, ten samples were collected using a sediment grab (0.1 m^2) at a site where intense aggregate extraction had ceased in 1997, and five at each of two adjacent non-extracted sites, all at 27–35 m depths. Invertebrates (1 mm) were identified, weighed, and counted. Data were combined with prior surveys undertaken using the same sampling design in 2001, 2002, 2003, 2004 and 2007.

(1) Desprez M. (2000) Physical and biological impact of marine aggregate extraction along the French coast of the Eastern English Channel: short-and long-term post-dredging restoration. *ICES Journal of Marine Science*, 57, 1428–1438.
(2) Simonini R., Ansaloni I., Bonini P., Grandi V., Graziosi F., Iotti M., Massamba-N'Siala G., Mauri M., Montanari G., Preti M. & De Nigris N. (2007) Recolonization and recovery dynamics of the macrozoobenthos after sand extraction in relict sand bottoms of the Northern Adriatic Sea. *Marine Environmental Research*, 64, 574–589.
(3) Vanaverbeke J. & Vincx M. (2008) Short-term changes in nematode communities from an abandoned intense sand extraction site on the Kwintebank (Belgian Continental Shelf) two years post-cessation. *Marine Environmental Research*, 66, 240–248.
(4) Foden J., Rogers S.I. & Jones A.P. (2009) Recovery rates of UK seabed habitats after cessation of aggregate extraction. *Marine Ecology Progress Series*, 390, 15–26.
(5) Bonne W.M. (2010) Macrobenthos characteristics and distribution, following intensive sand extraction from a subtidal sandbank. *Journal of Coastal Research*, 141–150.
(6) Barrio-Froján C.R., Cooper K.M., Bremner J., Defew E.C., Hussin W.M.W. & Paterson D.M. (2011) Assessing the recovery of functional diversity after sustained sediment screening at an aggregate dredging site in the North Sea. *Estuarine, Coastal and Shelf Science*, 92, 358–366.
(7) Waye-Barker G.A., McIlwaine P., Lozach S. & Cooper K.M. (2015) The effects of marine sand and gravel extraction on the sediment composition and macrofaunal community of a commercial dredging site (15 years post-dredging). *Marine Pollution Bulletin*, 99, 207–215.

4.16. Extract aggregates from a vessel that is moving rather than static

- **One study** examined the effects of dredging from a vessel that is moving rather than static on subtidal benthic invertebrate populations. The study was in the English Channel[1] (UK).

COMMUNITY RESPONSE (1 STUDY)

- **Overall species richness/diversity (1 study):** One site comparison study in the English Channel[1] found that a site where aggregate extraction was undertaken using a moving trailer suction hopper dredger had similar invertebrate species richness and lower diversity compared to a site where extraction occurred using a static suction hopper dredger.

POPULATION RESPONSE (1 STUDY)

- **Overall abundance (1 study):** One site comparison study in the English Channel[1] found that a site where aggregate extraction was undertaken using a moving trailer suction hopper dredger had higher abundance of invertebrates compared to a site where extraction occurred using a static suction hopper dredger.

Background

'Aggregates' is the collective term for sand, gravel and crushed rock. They are used as raw materials for the construction industry as well as for beach replenishment schemes. Aggregate extraction can impact subtidal benthic invertebrates through physical removal, loss and alteration of seabed and habitat, and direct physical damage from the machinery used (De Groot 1996). Two methods are commonly practised: anchor suction hopper dredging, a static type of extraction (De Groot 1996), and trailer suction hopper dredging which takes place from a moving vessel (Birchenough *et al.* 2010; Boyd & Rees 2003). In some areas, such as in the UK, both methods are used (Boyd & Rees 2003). Although both methods cause severe disturbance in seabed invertebrate communities (Desprez 2000, Sardá *et al*. 2000), the differences between dredged and surrounding undredged areas are more significant after static dredging (Boyd & Rees 2003). Trailer dredging is thought to reduce the intensity, and therefore the impact, of dredging in any one area by leaving small pockets of areas unaffected and from which recolonization and recovery may occur (Szymelfenig *et al*. 2006). Evidence for related interventions is summarised under "Threat: Energy production and mining – Mining and quarrying".

Birchenough S.N., Boyd S.E., Vanstaen K., Coggan R.A. & Limpenny D.S. (2010) Mapping an aggregate extraction site off the Eastern English Channel: a methodology in support of monitoring and management. *Estuarine, Coastal and Shelf Science*, 87, 420–430.

Boyd S.E. & Rees H.L. (2003) An examination of the spatial scale of impact on the marine benthos arising from marine aggregate extraction in the central English Channel. *Estuarine, Coastal and Shelf Science*, 57, 1–16.

De Groot S.J. (1996) The physical impact of marine aggregate extraction in the North Sea. *ICES Journal of Marine Science*, 53, 1051–1053.

Desprez M. (2000) Physical and biological impact of marine aggregate extraction along the French coast of the Eastern English Channel: short-and long-term post-dredging restoration. *ICES Journal of Marine Science,* 57, 1428–1438.

Sardá R., Pinedo S., Gremare A. & Taboada S. (2000) Changes in the dynamics of shallow sandy-bottom assemblages due to sand extraction in the Catalan Western Mediterranean Sea. *ICES Journal of Marine Science,* 57, 1446–1453.

Szymelfenig M., Kotwicki L. & Graca B. (2006) Benthic re-colonization in post-dredging pits in the Puck Bay (Southern Baltic Sea). *Estuarine, Coastal and Shelf Science*, 68, 489–498.

A site comparison study in 2000 of two soft seabed areas in the central English Channel, UK (1) found that using moving trailer rather than static suction hopper dredgers during aggregate extraction appeared to result in a similar number of invertebrate species, and a lower species diversity, but a higher abundance. Data were not statistically tested. The number of species at trailer- and static-dredged sites were similar (trailer: 20; static: 21). Species diversity was lower at the trailer dredged site than at the static dredged site (data presented as diversity indices). However, abundance of invertebrates was higher at the trailer dredged site (1,617 individuals/sample) compared to the static dredged site (103). In June 2000, sediment samples were collected using a sediment grab (0.1 m^2) from two sites at 18–25 m depths. One site had been dredged since 1968 by static suction, while the other had been dredged since 1989 by trailer suction. Invertebrates >0.5 mm were identified and counted from three to four samples/site.

(1) Boyd S.E. & Rees H.L. (2003) An examination of the spatial scale of impact on the marine benthos arising from marine aggregate extraction in the central English Channel. *Estuarine, Coastal and Shelf Science*, 57, 1–16.

4.17. Set limits for change in sediment particle size during aggregate extraction

- We found no studies that evaluated the effects of setting limits for change in sediment particle size during aggregate extraction on subtidal benthic invertebrate 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

'Aggregates' is the collective term for sand, gravel and crushed rock. They are used as raw materials for the construction industry as well as for beach replenishment schemes. During aggregate extraction, the unwanted part of the sediment is discarded back into the water or onto the seabed. Consequently, changes occur to the size of sediment particles, which can alter the natural seabed and the invertebrates living on or inside it (De Groot 1996). Limits for acceptable change in particle size during aggregate extraction can be set, with the aim of reducing the amount of alteration to seabed sediment properties. This may facilitate recovery following cessation of activities post-extraction (Cooper 2013). Additional evidence for intervention related to sediment discard during aggregate extraction and other activities are summarised under "Threat: Energy production and mining – Limit, cease, or prohibit sediment discard during aggregate extraction", "Remove discarded sediment material from the seabed following cessation of aggregate extraction", and "Set limits for change in sediment particle size during rock dump".

Cooper K.M. (2013) Setting limits for acceptable change in sediment particle size composition: Testing a new approach to managing marine aggregate dredging. *Marine Pollution Bulletin,* 73, 86–97.

Cooper K., Ware S., Vanstaen K. & Barry J. (2011) Gravel seeding - A suitable technique for restoring the seabed following marine aggregate dredging? *Estuarine, Coastal and Shelf Science*, 91, 121–132.

De Groot S.J. (1996). The physical impact of marine aggregate extraction in the North Sea. *ICES Journal of Marine Science*, 53, 1051–1053.

4.18. Limit, cease, or prohibit sediment discard during aggregate extraction

- We found no studies that evaluated the effects of limiting, ceasing, or prohibiting sediment discard during aggregate extraction on subtidal benthic invertebrate 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

'Aggregates' is the collective term for sand, gravel and crushed rock. They are used as raw materials for the construction industry as well as for beach replenishment schemes. During aggregate extraction, the unwanted part of the sediment is discarded back into the water or onto the seabed. Consequently, changes occur to the size of sediment particles, which can alter the natural seabed and the invertebrates living on or inside it, but additionally this discarded sediment portion can also directly physically impact invertebrates, for instance through smothering (De Groot 1996). Limiting, ceasing or prohibiting the discard of sediment during aggregate extraction can potentially reduce the amount of alteration to seabed sediment properties, and also avoid smothering of invertebrates. This may facilitate recovery following cessation of activities post-extraction (Cooper 2013). Additional evidence for intervention related to sediment discard during aggregate extraction and other activities are summarised under "Threat: Energy production and mining – Remove discarded sediment material from the seabed following cessation of aggregate extraction", and "Set limits for change in sediment particle size during rock dump".

Cooper K.M. (2013) Setting limits for acceptable change in sediment particle size composition: Testing a new approach to managing marine aggregate dredging. *Marine Pollution Bulletin,* 73, 86–97.
Cooper K., Ware S., Vanstaen K. & Barry J. (2011) Gravel seeding - A suitable technique for restoring the seabed following marine aggregate dredging? *Estuarine, Coastal and Shelf Science*, 91, 121–132.
De Groot S.J. (1996). The physical impact of marine aggregate extraction in the North Sea. *ICES Journal of Marine Science*, 53, 1051–1053.

4.19. Remove discarded sediment material from the seabed following cessation of aggregate extraction

- We found no studies that evaluated the effects of removing discarded sediment material from the seabed following cessation of aggregate extraction on subtidal benthic invertebrate 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

'Aggregates' is the collective term for sand, gravel and crushed rock. They are used as raw materials for the construction industry as well as for beach replenishment schemes (De Groot 1996). During extraction, a portion of sediment is often discarded, being unwanted or in excess depending on the industry requirements, and left on the seabed, changing its characteristics and further impacting invertebrates through habitat modifications. Following the cessation of aggregate extraction, this discarded material could be removed from the seabed through additional dredging (Cooper 2013), thereby removing the threat and potentially allowing natural recovery (Cooper *et al.* 2011). Additional evidence for intervention related to sediment discard during aggregate extraction and other activities are summarised under "Threat: Energy production and mining – Limit, cease, or prohibit sediment discard during aggregate extraction", and "Set limits for change in sediment particle size during rock dump".

Cooper K.M. (2013) Setting limits for acceptable change in sediment particle size composition: Testing a new approach to managing marine aggregate dredging. *Marine Pollution Bulletin,* 73, 86–97.
Cooper K., Ware S., Vanstaen K. & Barry J. (2011) Gravel seeding - A suitable technique for restoring the seabed following marine aggregate dredging? *Estuarine, Coastal and Shelf Science*, 91, 121–132.
De Groot S.J. (1996). The physical impact of marine aggregate extraction in the North Sea. *ICES Journal of Marine Science*, 53, 1051–1053.

4.20. Cease or prohibit marine mining

- **One study** examined the effects of ceasing or prohibiting mining on subtidal benthic invertebrate populations. The study was in the Bering Sea[1] (USA).

COMMUNITY RESPONSE (1 STUDY)

- **Overall community composition (1 study):** One site comparison study in the Bering Sea[1] found that following cessation of gold mining, overall invertebrate community composition became similar to that of an unmined site.
- **Overall richness/diversity (1 study):** One site comparison study in the Bering Sea[1] found that following cessation of gold mining, overall invertebrate richness and diversity became similar to that of an unmined site.

POPULATION RESPONSE (1 STUDY)

- **Overall abundance (1 study):** One site comparison study in the Bering Sea[1] found that following cessation of gold mining, overall invertebrate abundance and biomass became similar to that of an unmined site.

Background

Marine mining involves the retrieval of minerals from the seabed, usually through dredging. While certain minerals mostly occur in the deep sea, where mining is yet to be undertaken at an industrial scale, offshore and coastal mining does occur in parts of the world (Miller *et al*. 2018). Mining can have negative impacts on subtidal benthic invertebrates through physical disturbances from dredging operations, chemical contamination, and changes in sediment characteristics (Jewett *et al*. 1999). Mining could be ceased (for instance following end of licence or voluntary cessation of activities) or prohibited (through legislation) in an area, and the site and its invertebrates left to naturally recover over time (Jewett *et a*l. 1999).

Jewett S.C., Feder H.M. & Blanchard A. (1999) Assessment of the benthic environment following offshore placer gold mining in the northeastern Bering Sea. *Marine Environmental Research*, 48, 91–122.

Miller K.A., Thompson K.F., Johnston P. & Santillo D. (2018) An overview of seabed mining including the current state of development, environmental impacts, and knowledge gaps. *Frontiers in Marine Science*, 4, 418.

A site comparison study in 1986–1993 of two sites of mixed seabed in the northeastern Bering Sea, Alaska, USA (1) found that ceasing gold mining at a site led to invertebrate community composition, biomass, abundance, taxa richness and diversity becoming similar to that of an unmined site, after three to five years depending on the sediment type. Community composition at the mined site had become more similar to that of the unmined site after four to five years in sandy sediments, and three years in cobbly sediments (presented as graphical analyses). In sands, invertebrate biomasses were similar to unmined sites after four years, and abundances, number of taxa and diversities were similar after five years (richness mined: 27, unmined: 33; see study for biomass, abundance and diversity data). In cobbles after three years, mined and unmined sites had similar invertebrate biomasses, abundances, number of taxa (mined: 29, unmined: 39), and diversities. An area was mined for gold in June–November 1986. Yearly in 1987–1991 and in 1993, one site in the mined area and one unmined site approximately 10 km away were surveyed. Each site had areas of sandy and areas of cobbly sediments. During each survey, divers collected three samples/sediment type/site using a suction sampler (0.1 m^2, 10 cm depth). Invertebrates (>1 mm) were identified, counted and wet-weighed.

(1) Jewett S.C., Feder H.M. & Blanchard A. (1999) Assessment of the benthic environment following offshore placer gold mining in the northeastern Bering Sea. *Marine Environmental Research*, 48, 91–122.

4.21. Cease or prohibit mining waste (tailings) disposal at sea

• We found no studies that evaluated the effects of ceasing or prohibiting mining waste (tailings) disposal at sea on subtidal benthic invertebrate 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

Mine tailings (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 the marine environment. These mine tailings disposals are known as "submarine tailing disposal" in shallow waters, and "deep sea tailings disposal" in deeper waters (Vare *et al.* 2018). Mine tailings can have negative impacts on subtidal benthic invertebrates through physical disturbances, smothering, chemical contamination (Marinho *et al*. 2017), and changes in sediment characteristics (Kathman *et al.* 1983) and submarine tailing disposal has been prohibited in parts of the world (Kline & Stekoll 2001). Ceasing or prohibiting submarine mine tailings disposal in an area can remove the source of harm and potentially allow natural recovery of the seabed and its invertebrate community. Evidence for interventions related to marine mining are summarised under "Threat: Energy production and mining – Cease or prohibit marine mining" and "Leave mine tailings in place following cessation of activities".

Kathman R.D., Brinkhurst R.O., Woods R.E. & Jeffries D.C. (1983) *Benthic studies in Alice Arm and Hastings Arm, BC in relation to mine tailings dispersal*. Institute of Ocean Sciences, Department of Fisheries and Oceans.

Kline E.R. & Stekoll M.S. (2001) Colonization of mine tailings by marine invertebrates. *Marine Environmental Research*, 51, 301–325.

Marinho C.H., Giarratano E., Esteves J.L., Narvarte M.A. & Gil M.N. (2017) Hazardous metal pollution in a protected coastal area from Northern Patagonia (Argentina). *Environmental Science and Pollution Research*, 24, 6724–6735.

Vare L.L., Baker M.C., Howe J.A., Levin L.A., Neira C., Ramirez-Llodra E.Z., Reichelt-Brushett A., Rowden A.A., Shimmield T.M., Simpson S.L. & Soto E.H. (2018). Scientific considerations for the assessment and management of mine tailings disposal in the deep sea. *Frontiers in Marine Science*, 5, 17.

4.22. Leave mining waste (tailings) in place following cessation of disposal operations

• **One study** examined the effects of leaving mining waste (tailings) in place following cessation of disposal operations on subtidal benthic invertebrate populations. The study was in Auke Bay[1] (USA).

COMMUNITY RESPONSE (1 STUDY)

- **Overall community composition (1 study):** One replicated, paired, controlled study in Auke Bay[1] found that plots where mine tailings were left in place had similar invertebrate community composition as plots where tailings had been removed, but both had different communities to plots of natural sediment.
- **Overall richness/diversity (1 study):** One replicated, paired, controlled study in Auke Bay[1] found that plots where mine tailings were left in place had similar invertebrate species richness

as plots where tailings had been removed, but both had lower richness compared to plots of natural sediment.

POPULATION RESPONSE (1 STUDY)

- **Overall abundance (1 study):** One replicated, paired, controlled study in Auke Bay[1] found that plots where mine tailings were left in place had similar invertebrate overall abundance and biomass as plots where tailings had been removed. While plots with and without tailings had similar abundances to plots of natural sediment, their biomasses were higher.

Background

Mine tailings (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 the marine environment. These mine tailings disposals are known as "submarine tailing disposal" in shallow waters, and "deep sea tailings disposal" in deeper waters (Vare *et al.* 2018). Mine tailings can have negative impacts on subtidal benthic invertebrates through physical disturbances, smothering, chemical contamination (Marinho *et al*. 2017), and changes in sediment characteristics (Kathman *et al.* 1983) and submarine tailing disposal has been prohibited in parts of the world (Kline & Stekoll 2001). However, where it occurs, following cessation of activities, removal can incur additional disturbances. As such, leaving mine tailings in place following cessation of activities, and allowing the potential natural recovery of the seabed and its invertebrate community (Kline & Stekoll 2001), can perhaps reduce the risk of additional impacts resulting from their removal. Evidence for related interventions is summarised under "Threat: Energy production and mining – Cease or prohibit submarine mining disposal".

Kathman R.D., Brinkhurst R.O., Woods R.E. & Jeffries D.C. (1983) *Benthic studies in Alice Arm and Hastings Arm, BC in relation to mine tailings dispersal*. Institute of Ocean Sciences, Department of Fisheries and Oceans.

Kline E.R. & Stekoll M.S. (2001) Colonization of mine tailings by marine invertebrates. *Marine Environmental Research*, 51, 301–325.

Marinho C.H., Giarratano E., Esteves J.L., Narvarte M.A. & Gil M.N. (2017) Hazardous metal pollution in a protected coastal area from Northern Patagonia (Argentina). *Environmental Science and Pollution Research*, 24, 6724–6735.

Vare L.L., Baker M.C., Howe J.A., Levin L.A., Neira C., Ramirez-Llodra E.Z., Reichelt-Brushett A., Rowden A.A., Shimmield T.M., Simpson S.L. & Soto E.H. (2018). Scientific considerations for the assessment and management of mine tailings disposal in the deep sea. *Frontiers in Marine Science*, 5, 17.

A replicated, paired, controlled, pilot study in 1994–1996 of 90 plots of soft seabed in Auke Bay, Alaska, USA (1) found that leaving mine tailings on the seabed after ceasing disposal operations, or removing them, led to similar changes in invertebrate community composition, abundance, biomass and species richness, but either way remained different to nearby natural communities, after 22 months. After 22 months, invertebrate community compositions were similar in plots with and without tailings but remained different to plots of natural sediment (data presented as graphical analyses). Plots with and without tailings had similar invertebrate abundance (with: 900 vs without: 1,050 individuals/tray), biomass (370 vs 380 mg/tray), and species richness (50 vs 48 species/tray). Plots with and without tailings had similar abundances to the natural plot (natural plot abundance: 920 individuals/tray), but their biomasses were higher (natural plot biomass: 150 mg/tray,) and richness were lower (natural plot species richness: 40 species/tray). In 1994, 48 plastic trays (as experimental plots, 8 cm deep, 15 cm diameter) were filled with either tailings or sediments without invertebrates (to mimic removal of tailings) and deployed in pairs by divers at 21 m depth in a circular arrangement (30 m diameter). After 9, 17, and 22 months, 10 trays/treatment were recovered (in total: 30 of the 48 trays), and 10 plots of nearby natural sediment were sampled using a tray as a corer. Invertebrates (>500 μm) were identified, counted, and dry-weighed.

(1) Kline E.R. & Stekoll M.S. (2001) Colonization of mine tailings by marine invertebrates. *Marine Environmental Research*, 51, 301–325.

Renewable energy

4.23. Limit the number and/or extent of, or prohibit additional, renewable energy installations in an area

- We found no studies that evaluated the effects of limiting the number and/or extent of, or prohibit additional, renewable energy installations in an area on subtidal benthic invertebrate 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

Renewable energy installations, such as marine wind farms, are becoming widespread in the marine environment (Boehlert & Gill 2010). However, their occurrence can negatively impact subtidal benthic invertebrates through direct physical damage when the infrastructure is constructed, changes in hydrology, and loss of habitat (from sedimentary grounds to hard surfaces) (Langhamer 2012). The number of renewable energy installations, or their spatial extent (area of seabed covered), could be limited in one area, for instance through the development of marine protected areas, bylaws, or other legislation. Doing so could limit the area impacted by renewable energy installations and the intensity of the pressure, thereby limiting the negative impacts on subtidal benthic invertebrates. Evidence for intervention related to marine spatial planning and co-location of activities are summarised under "Threat: Energy production and mining – Co-locate aquaculture systems with other activities and other infrastructures (such as wind farms) to maximise use of marine space".

Boehlert G. & Gill A. (2010) Environmental and ecological effects of ocean renewable energy development: A current synthesis. *Oceanography*, 23, 68–81.
Langhamer O. (2012) Artificial reef effect in relation to offshore renewable energy conversion: State of the art. *The Scientific World Journal*.

4.24. Co-locate aquaculture systems with other activities and other infrastructures (such as wind farms) to maximise use of marine space

- We found no studies that evaluated the effects of limiting the number and/or extent of, or prohibit additional, renewable energy installations in an area on subtidal benthic invertebrate 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

Aquaculture systems can negatively impact invertebrate subtidal communities through damages to the seabed, pollution, or spread of non-native species (Wu *et al.* 1994). Some of these threats are also commonly associated with other anthropogenic activities or infrastructures, such as oil rigs and wind farms (Gimpel *et al.* 2015; Inger *et al*. 2009). By co-locating aquaculture systems with these activities and infrastructure, the cumulative negative impacts can be spatially limited and constrained in extent, therefore potentially preventing their occurrence elsewhere, or allowing recovery in the case of relocation. Marine spatial planning can help with identifying suitable area for the occurrence of multiple complex activities (Douvere 2008; Inger *et al*. 2009). Evidence for interventions related to aquaculture relocation are summarised under "Threat: Pollution – Locate aquaculture systems in already impacted areas", "Locate artificial reefs near aquaculture systems to act as biofilters", and "Habitat restoration and creation – Locate artificial reefs near aquaculture systems to benefit from nutrient run-offs".

Douvere F. (2008) The importance of marine spatial planning in advancing ecosystem-based sea use management. *Marine Policy*, 32, 762–771.

Gimpel A., Stelzenmüller V., Grote B., Buck B.H., Floeter J., Núñez-Riboni I., Pogoda B. & Temming A. (2015) A GIS modelling framework to evaluate marine spatial planning scenarios: Co-location of offshore wind farms and aquaculture in the German EEZ. *Marine Policy,* 55, 102–115.

Inger R., Attrill M.J., Bearhop S., Broderick A.C., Grecian W.J., Hodgson D.J., Mills C., Sheehan E., Votier S.C., Witt M.J. & Godley B.J. (2009) Marine renewable energy: potential benefits to biodiversity? An urgent call for research. *Journal of Applied Ecology*, 46,1145–1153.

Wu R.S.S., Lam K.S., MacKay D.W., Lau T.C. & Yam V. (1994) Impact of marine fish farming on water quality and bottom sediment: A case study in the sub-tropical environment. *Marine Environmental Research,* 38, 115–145.

5. Threat: Transportation and service corridors

Background

Threats from transportation and service corridors include infrastructures such as ships and shipping lanes, ferries and bridges, communication and power cables, and oil and gas pipelines, and associated threats from their activities.

The greatest threats to subtidal benthic invertebrates from transportation and service corridors tend to be from the destruction and pollution of habitats (Chou 2006; Waldock *et al*. 1988), due for instance to scouring, anchoring damages, leaching of chemicals from the hull of ships, or the disposal of wastes and garbage from vessels. Interventions in response to these threats are described in other chapters: "Habitat restoration and creation" and "Threat: Pollution".

An increasingly important threat relates to the introduction and spread of non-native, invasive or problematic species due to transportation and service corridors, for example either on the hull of ships, in ballast waters, or inside aquaculture trade products (Hulme, 2009). Interventions related to the introduction and spread of non-native, invasive or problematic species are described in "Threat: Non-native, invasive and problematic species".

Other interventions related to transportation and service corridors are discussed below. Note that pre-emptive actions aiming to prevent the occurrence of a threat at one location (e.g. "prevent cable routings from going through sensitive areas") are not described here, as robustly testing for their effect would not be feasible. However, pre-emptive management actions that can be undertaken at the planning stage before an activity takes place, and aiming to reduce the likelihood or level of a threat (e.g. "Use cables of smaller width") are included in this chapter.

Chou L.M. (2006) Marine Habitats in One of the World's Busiest Harbours. In: Wolanski E. (eds) *The Environment in Asia Pacific Harbours*. Springer, Dordrecht.

Hulme P.E. (2009) Trade, transport and trouble: managing invasive species pathways in an era of globalization. *Journal of Applied Ecology*, 46, 10–18.

Waldock M.J., Waite M.E. & Thain J.E. (1988) Inputs of tbt to the marine environment from shipping activity in the U.K., *Environmental Technology Letters,* 9, 999–1010.

Utility and service lines

5.1. Set limits on the area that can be covered by utility and service lines at one location

- We found no studies that evaluated the effects of setting limits on the area that can be covered by utility and service lines at one location on subtidal benthic invertebrate 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

Utility and service lines, such as communication and power cables, and oil and gas pipelines, can impact subtidal benthic invertebrates through physical damage and habitat loss. Limits could be set on the area of seabed that can be covered by utility and service lines. This may reduce the level and spatial extent of threat by preventing additional installation, and therefore benefit subtidal benthic invertebrates.

5.2. Use cables and pipelines of smaller width

- We found no studies that evaluated the effects of using cables and pipelines of smaller width on subtidal benthic invertebrate 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

Communication and power cables, and oil and gas pipelines, can impact subtidal benthic invertebrates through physical damage and habitat loss. Limits on the width of cables and pipelines could be set at the planning stage before laying them on the seabed. This may reduce the level of threat by limiting the extent of damage, and potentially benefit subtidal benthic invertebrates.

5.3. Bury cables and pipelines in the seabed rather than laying them on the seabed

- We found no studies that evaluated the effects of burying cables and pipelines in the seabed rather than laying them on the seabed on subtidal benthic invertebrate 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

Communication and power cables, and oil and gas pipelines, can impact subtidal benthic invertebrates through physical damage and habitat loss. The burial process could be planned ahead of installation, or at a later stage following the lay-out of the cable or pipeline, with the aim of reducing physical impacts on the seabed and on subtidal benthic invertebrates (Kraus & Carter 2018). This may allow natural sediment to cover the cables or pipelines, thereby recreating a suitable habitat, and potentially allowing recolonization by subtidal benthic invertebrates. Evidence related to the burial of pipelines instead of rock dumping are summarised under "Threat: Energy production and mining – Bury pipelines instead of surface laying and rock dumping". Evidence related to promoting biodiversity around subsea cables and pipelines are summarised under "Habitat restoration and creation – Cover subsea cables with artificial reefs" and "Cover subsea cables with materials that encourage the accumulation of natural sediments".

Kraus C. & Carter L. (2018) Seabed recovery following protective burial of subsea cables - Observations from the continental margin. *Ocean Engineering,* 157, 251–261.

5.4. Use a different technique when laying and burying cables and pipelines

- We found no studies that evaluated the effects of using a different technique when laying and burying cables and pipelines on subtidal benthic invertebrate 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

Communication and power cables, and oil and gas pipelines, can impact subtidal benthic invertebrates through physical damage and habitat loss. The burial process could be planned ahead of installation with the aim of reducing physical impacts on the seabed and on subtidal benthic invertebrates. For instance, using ploughing techniques rather than water-jetted trenching or directional drilling can be less disruptive to subtidal benthic invertebrates and promote recovery following disturbance (Kraus & Carter 2018).

Kraus C. & Carter L. (2018) Seabed recovery following protective burial of subsea cables - Observations from the continental margin. *Ocean Engineering,* 157, 251–261.

5.5. Remove utility and service lines after decommissioning

- We found no studies that evaluated the effects of removing utility and service lines after decommissioning on subtidal benthic invertebrate 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

Utility and service lines can impact subtidal benthic invertebrates through physical damage and habitat loss. Along with associated infrastructures (Brigitte *et al*. 2018), it may be possible to remove utility and service lines after decommissioning (Ekins *et al*. 2006). This may remove the threat and allow for recolonization and recovery over time. Evidence related to the potential benefits of leaving pipelines in places are summarised under "Threat: Transportation and service corridors – Leave utility and service lines in place after decommissioning".

Brigitte S., Fowler A.M., Macreadie P.I., Palandro D.A., Aziz A.C. & Booth D.J. (2018) Decommissioning of offshore oil and gas structures–Environmental opportunities and challenges. *Science of the Total Environment*, 658, 973–981.
Ekins P., Vanner R. & Firebrace J. (2006) Decommissioning of offshore oil and gas facilities: A comparative assessment of different scenarios. *Journal of Environmental Management*, 79, 420–438.

5.6. Leave utility and service lines in place after decommissioning

- We found no studies that evaluated the effects of leaving utility and service lines in place after decommissioning on subtidal benthic invertebrate 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

Utility and service lines can impact subtidal benthic invertebrates through physical damage and habitat loss. Along with associated infrastructures (Brigitte *et al.* 2018), it may be possible to leave utility and service lines in place after decommissioning (Ekins *et al.* 2006). This may avoid the threats associated with disturbance and physical damage from the removal of these structures. Evidence related to the potential benefits of removing pipelines in places are summarised under "Threat: Transportation and service corridors – Remove utility and service lines after decommissioning". Evidence related to promoting biodiversity around subsea cables and pipelines are summarised under "Habitat restoration and creation – Cover subsea cables with artificial reefs" and "Cover subsea cables with materials that encourage the accumulation of natural sediments".

Brigitte S., Fowler A.M., Macreadie P.I., Palandro D.A., Aziz A.C. & Booth D.J. (2018) Decommissioning of offshore oil and gas structures–Environmental opportunities and challenges. *Science of the Total Environment*, 658, 973–981.

Ekins P., Vanner R. & Firebrace J. (2006) Decommissioning of offshore oil and gas facilities: A comparative assessment of different scenarios. *Journal of Environmental Management*, 79, 420–438.

Shipping lanes

5.7. Cease or prohibit shipping

- **Three studies** examined the effects of ceasing or prohibiting shipping on subtidal benthic invertebrate populations. All studies were in the North Sea[1–3] (Belgium, Germany, Netherlands).

COMMUNITY RESPONSE (2 STUDIES)

- **Overall community composition (1 study):** One site comparison study in the North Sea[2] found that areas closed to shipping developed different overall invertebrate community compositions compared to areas where shipping occurred.
- **Overall species richness/diversity (1 study):** One site comparison study in the North Sea[2] found that areas closed to shipping did not develop different overall invertebrate species richness and diversity compared to areas where shipping occurred.

POPULATION RESPONSE (2 STUDIES)

- **Overall abundance (2 studies):** Two site comparison studies (one before-and-after) in the North Sea[2,3] found that areas closed to shipping had similar overall invertebrate abundance and biomass compared to areas where shipping occurred.
- **Overall abundance (2 studies):** Two site comparison studies (one before-and-after) in the North Sea[2,3] found that areas closed to shipping had similar overall invertebrate abundance and biomass compared to areas where shipping occurred.

OTHER (2 STUDIES)

- **Overall community energy flow (1 study):** One before-after, site comparison study in the North Sea[1] found that after closing an area to shipping, invertebrate community energy flow did not change, but it increased in nearby areas where shipping occurred.
- **Species energy flow (1 study):** One before-and-after, site comparison study in the North Sea[1] found that closing an area to shipping had mixed effects on species-level energy flow.

Background

Shipping (here meaning the movement of any commercial vessels, including cargo ships but also fishing ships) can impact subtidal benthic invertebrates through physical disturbance, introduction of non-native species and pollution. When shipping is undertaken for fishing, it can also lead to additional pressure from the fishing activities. Shipping can be stopped or prohibited in specific areas. For instance, buffer zones around windfarm structures or gas platforms are usually set and exclude any ship from entering (Coates *et al.* 2016). In these instances, shipping closures also acts as fisheries closures. These exclusion zones can either be temporary, such as during the windfarm construction (Roach *et al*. 2018), or permanent (Bergman *et al.* 2015). Ceasing shipping activities can potentially benefit subtidal benthic invertebrates by reducing and/or removing this pressure, allowing them to potentially recover over time. A direct consequence of ceasing shipping is the cessation of fishing, and in particular bottom trawling, for which evidence has been summarised under "Threat: Biological Resource Use – Cease or prohibit bottom trawling". Evidence for related interventions are summarised under "Threat: Biological resource use – Cease or prohibit all types of fishing" and "Establish temporary fisheries closure", as well as under "Habitat restoration and creation – Place anthropogenic installations (e.g. windfarms) in an area to act as artificial reefs and reduce the level of fishing". Evidence related to the creation of Particularly Sensitive Sea Areas which regulate and manage shipping (IMO Resolution A.982(24)) is summarised under "Habitat protection – Designate a Particularly Sensitive Sea Area (PSSA) to regulate impactful maritime activities".

Bergman M.J.N., Ubels S.M., Duineveld G.C.A. & Meesters E.W.G. (2015) Effects of a 5-year trawling ban on the local benthic community in a wind farm in the Dutch coastal zone. *ICES Journal of Marine Science*, 72, 962–972.

Coates D.A., Kapasakali D.A., Vincx M. & Vanaverbeke J. (2016) Short-term effects of fishery exclusion in offshore wind farms on macrofaunal communities in the Belgian part of the North Sea. *Fisheries Research*, 179, 131–138.

IMO Assembly Resolution 24/982 (2005) Revised guidelines for the identification and designation of Particularly Sensitive Sea Areas.

Roach M., Cohen M., Forster R., Revill A.S., Johnson M. & Handling editor: Steven Degraer. (2018) The effects of temporary exclusion of activity due to wind farm construction on a lobster (*Homarus gammarus*) fishery suggests a potential management approach. *ICES Journal of Marine Science*, 75, 1416–1426.

A before-after, site comparison study in 2003–2004 in areas of soft seabed sediment in the German Bight, southern North Sea, Germany (1) found that, during the 12–14 months after closing an area to shipping, community energy flow (related to community structure) at sites within the closed area did not change, but it increased in nearby open sites where shipping occurred. Before shipping closure, community energy flow was similar in the closed (80 kJ/m^2) and open sites (66 kJ/m^2). After 12–14 months, community energy flow in the closed sites was similar to before (69 kJ/m^2), but lower than at open sites where energy flow had increased over time (92 kJ/m^2). After 12 months, species-level energy flow was higher in closed areas compared to open areas for 10 of 70 species, and lower for 7 of 70 species. In July 2003 a pilot windfarm platform was constructed, and the surrounding area (500 m radius) closed to all shipping (including fishing vessels). Invertebrates were surveyed at 10 sites inside the windfarm area and 10–18 outside (9 km away) before construction (March–August), and 12–14 months after exclusion (July–October 2004). Invertebrates were collected using a sediment grab (0.1 m^2) and a beam trawl at 28 m depth. All were identified, counted, weighed, and their biomass converted to energy values (kilo Joule) using conversion factors. Energy flow was used to compare communities.

A before-and-after, site comparison study in 2008–2012 of multiple sites in an area of sandy seabed in the southern North Sea, 40–50 km off the coast of Belgium (2) found that the three years after closing an area to shipping, overall community composition was different in closed and open sites where shipping occurred, but total abundance, biomass, species richness and diversity remained similar across sites. Data and analyses of community compositions were not reported. Total invertebrate abundance did not change over time and remained similar at sites closed and open to shipping, before (2008: closed 361 vs open 436 individuals/m^2) and one to two years after the closure (2011–2012: 369–1,027 vs 256–458). This was also true for total biomass (2008: 802 vs 1,656; 2011–2012: 514–5,733 vs 1,392–1,864 mg/m^2), species richness (2008: 10.3 vs 10.7; 2011–2012: 10.4–12.3 vs 10.3–14.7 species/sample), and diversity (reported as diversity index). In 2009-2010 a windfarm was constructed, and an area of approximately 21 km^2 closed to all shipping (including fishing vessels) was established around the windfarm (500 m radius). Invertebrates were surveyed at 6–16 sites inside the windfarm area and 15–25 outside before construction in 2008, and after in 2011 and 2012 (always in September–October). Invertebrates >1 mm were collected using a sediment grab (0.1 m^2) at 15–40 m depth, identified, counted, and their dried biomass measured or estimated.

A site comparison study in 2011 of seven areas of soft seabed in the southern North Sea, Netherlands (3) found that overall, an area closed to shipping had similar invertebrate abundance, biomass, species richness and diversity, compared to six adjacent open areas where shipping occurred, after five years. For each metric, not all data were shown. From core samples, all areas had similar invertebrate abundance (min. 1,096/m^2; (closed area); max. 1,778/m^2 (open area)), biomass (min. 32 g/m^2 (closed area); max. 17 g/m^2 (open area)), number of species (closed: 16; open: 13–20), and diversity (as diversity indices). From dredge samples, invertebrate abundance and species diversity were similar in the closed area and five of six open areas, while all areas had similar biomass (min. 61 g/m^2 (closed area); max. 134 g/m^2 (closed area)) and number of species (closed: 20; open: 15–21). An offshore wind farm was constructed in 2006, with a 500 m buffer zone (approximately 25 km^2) around it closed to all shipping (including fishing vessels). Invertebrates inside the closed area and at six nearby open areas were surveyed in February 2011 using two methods. Shorter-lived infauna (>1 mm) were sampled using sediment core (0.078 m^2; 16 samples across the closed area; 8 samples/open areas). Longer-lived infauna and epifauna (>7 mm) were sampled using a dredge (20 m^2; 14 samples across the closed area; 6 samples/open areas). All invertebrates were identified, counted, and weighed (results are for dry weights).

(1) Dannheim J., Brey T., Schröder A., Mintenbeck K., Knust R. & Arntz W.E. (2014) Trophic look at soft-bottom communities — Short-term effects of trawling cessation on benthos. *Journal of Sea Research,* 85, 18–28.

(2) Coates D.A., Kapasakali D.A., Vincx M. & Vanaverbeke J. (2016) Short-term effects of fishery exclusion in offshore wind farms on macrofaunal communities in the Belgian part of the North Sea. *Fisheries Research,* 179, 131–138.

(3) Bergman M.J.N., Ubels S.M., Duineveld G.C.A. & Meesters E.W.G. (2015) Effects of a 5-year trawling ban on the local benthic community in a wind farm in the Dutch coastal zone. *ICES Journal of Marine Science,* 72, 962–972.

5.8. Divert shipping routes

- We found no studies that evaluated the effects of diverting shipping routes on subtidal benthic invertebrate 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

Shipping can impact subtidal benthic invertebrates through physical disturbance, introduction of non-native species and pollution. Shipping routes could be diverted, either permanently or temporarily, to reduce the level of impact at one location, and allow for natural recovery. This may potentially benefit subtidal benthic invertebrates through reduced disturbance.

5.9. Limit, cease or prohibit recreational boating

- We found no studies that evaluated the effects of limiting, ceasing or prohibiting recreational fishing and/or harvesting on subtidal benthic invertebrate 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

Recreational boating, including powerboats, sailboats, or man-powered vessels such as rowing and paddle boats, can impact subtidal benthic invertebrates through physical damage from anchors and propellers (Lloret *et al.* 2008; Milazzo *et al.* 2002; Whitfield *et al*. 2002), disturbance from fast flowing water or suspension of sediments, or pollution (Hammerstrom *et al.* 2007). Recreational boating could be limited in one area, by restricting the activity in space and time (limits on duration and occurrence, delimiting allowed areas). Boating could also be ceased by setting a permanent or temporary closure (e.g. seasonal closure), or prohibited through bylaws. This may help reduce the intensity of the threats associated with these activities and potentially allow subtidal benthic invertebrate communities to persist or recover over time. When restrictions of recreational boating occur in the context of a marine protected area, evidence is summarised under "Habitat protection", including "Habitat protection - Designate a Marine Protected Area and set a no-anchoring zone".

Hammerstrom K.K., Kenworthy W.J., Whitfield P.E. & Merello M.F. (2007) Response and recovery dynamics of seagrasses *Thalassia testudinum* and *Syringodium filiforme* and macroalgae in experimental motor vessel disturbances. *Marine Ecology Progress Series* 345,83–92.

Lloret J., Zaragoza N., Caballero D. & Riera V. (2008) Impacts of recreational boating on the marine environment of Cap de Creus (Mediterranean Sea). *Ocean & Coastal Management,* 51, 749–754.

Milazzo M., Chemello R., Badalamenti F., Camarda R. & Riggio S. (2002) The impact of human recreational activities in Marine Protected Areas: What lessons should be learnt in the Mediterranean Sea? *Marine Ecology,* 23, 280–290.

Whitfield P., Kenworthy W., Hammerstrom K. & Fonseca M. (2002) The Role of a Hurricane in the Expansion of Disturbances Initiated by Motor Vessels on Seagrass Banks. *Journal of Coastal Research*, 86–99.

5.10.Limit, cease or prohibit anchoring from ships/boats/vessels

- We found no studies that evaluated the effects of setting limits, ceasing or prohibitinganchroing from ships/boats/vessels on subtidal benthic invertebrate 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

Anchors are designed to dig into or hook onto the seabed to secure a vessel's position and prevent it from drifting with the winds or currents. Anchoring of recreational and/or commercial vessels can impact subtidal benthic invertebrates through physical damage, disturbance to and penetration into the seabed from anchors and chains (Griffith *et al*. 2017; Lloret *et al.* 2008; Whitfield *et al*. 2002). Structurally complex seabed habitats, such as seagrass and mussel beds, or oyster and coral reels, are considered particularly at risks from recreational anchoring (Hammerstrom *et al.* 2007). The areas where vessels (commercial or recreational) are allowed to anchor could be limited or ceased, for instance by restricting the activity in space and time (limits on duration and occurrence) or restricting the number of anchors allowed at any one time. Anchoring could also be ceased by setting a permanent or temporary closure (e.g. seasonal closure), or prohibited through bylaws and the creation of no-anchoring zones (Griffith *et al*. 2017). This may help reduce the intensity of the threats associated with anchoring and potentially allow subtidal benthic invertebrate communities to persist or recover over time.

For evidence related to recreational anchoring within a marine protected area, see "Habitat protection – Designate a Marine Protected Area and set a no-anchoring zone". Evidence for other interventions related to anchoring is summarised under "Threat: Transportation and service corridors – Use a different type of anchor" and "Provide additional moorings to reduce anchoring".

Griffiths C.A., Langmead O.A., Readman J.A.J. & Tillin H.M. (2017) *Anchoring and Mooring Impacts in English and Welsh Marine Protected Areas: Reviewing sensitivity, activity, risk and management.* A report to Defra Impacts Evidence Group.
Hammerstrom K.K., Kenworthy W.J., Whitfield P.E. & Merello M.F. (2007) Response and recovery dynamics of seagrasses *Thalassia testudinum* and *Syringodium filiforme* and macroalgae in experimental motor vessel disturbances. *Marine Ecology Progress Series* 345,83–92.
Lloret J., Zaragoza N., Caballero D. & Riera V. (2008) Impacts of recreational boating on the marine environment of Cap de Creus (Mediterranean Sea). *Ocean & Coastal Management*, 51, 749–754.
Whitfield P., Kenworthy W., Hammerstrom K. & Fonseca M. (2002) The role of a hurricane in the expansion of disturbances initiated by motor vessels on seagrass banks. *Journal of Coastal Research*, 86–99.

5.11.Use a different type of anchor

- We found no studies that evaluated the effects of using a different type of anchor on subtidal benthic invertebrate 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

Anchors are designed to dig into or hook onto the seabed to secure a vessel's position and prevent it from drifting with the winds or currents. Anchoring can impact subtidal benthic invertebrates through physical damage, disturbance to and penetration in the seabed (Griffiths *et al*. 2017). Because anchor size, weight and design affect the level of impact caused (Liley *et al.* 2012), a different type of anchors, such as sacrificial anchors, could be used to reduce the level of impact caused. For recreational boats, the use of a particular type of anchor has been shown to be effective in reducing impact in seagrass beds (Milazzo *et al.* 2004). Evidence for other interventions related to anchoring are summarised under "Threat: Transportation and service corridors – Provide additional moorings to reduce anchoring", and "Set limits or reduce the area where ships/boats/vessels can anchor".

Griffiths C.A., Langmead O.A., Readman J.A.J. & Tillin H.M. (2017) *Anchoring and Mooring Impacts in English and Welsh Marine Protected Areas: Reviewing sensitivity, activity, risk and management.* A report to Defra Impacts Evidence Group.

Liley D., Morris R.K.A., Cruickshanks K., Macleod C., Underhill-Day J., Brereton T. & Mitchell J., (2012) *Identifying best practice in management of activities on Marine Protected Areas.* Footprint Ecology/Bright Angel Consultants/MARINElife. Natural England Commissioned Reports, Number 108.

Milazzo M., Badalamenti F., Ceccherelli G. & Chemello R. (2004) Boat anchoring on *Posidonia oceanica* beds in a marine protected area (Italy, western Mediterranean): effect of anchor types in different anchoring stages. *Journal of Experimental Marine Biology and Ecology*, 299, 51–62.

5.12. Provide additional moorings to reduce anchoring

- We found no studies that evaluated the effects of providing additional moorings to reduce anchoring on subtidal benthic invertebrate 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

Anchoring can impact subtidal benthic invertebrates through physical damage, disturbance to and penetration in the seabed (Griffiths *et al*. 2017). Moorings, permanent or temporary, provide an alternative to anchoring and allow for reduced anchoring in the area. Moorings can be provided for recreational vessels, as well as commercial vessels up to specific sizes. The damaging impact to the seabed is contained within the mooring vicinity rather than at multiple anchoring sites. Evidence for other interventions related to anchoring are summarised under "Threat: Transportation and service corridors – Use moorings which reduce or avoid contact with the seabed (eco-moorings)", "Use a different type of anchor", and "Set limits or reduce the area where ships/boats/vessels can anchor".

Griffiths C.A., Langmead O.A., Readman J.A.J. & Tillin H.M. (2017) *Anchoring and Mooring Impacts in English and Welsh Marine Protected Areas: Reviewing sensitivity, activity, risk and management.* A report to Defra Impacts Evidence Group.

5.13. Use moorings which reduce or avoid contact with the seabed (eco-moorings)

- We found no studies that evaluated the effects of using moorings which reduce or avoid contact with the seabed (eco-moorings) on subtidal benthic invertebrate 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

Moorings, permanent or temporary, provide an alternative to anchoring and allow for reduced anchoring in the area. Moorings can be provided for recreational vessels, as well as commercial vessels up to specific sizes. However, moorings also impact subtidal benthic invertebrates through chronic damage and physical disturbance to the seabed (Griffiths *et al*. 2017). Moorings that reduce or avoid contact with the seabed, known as eco-moorings, could be used to reduce disturbance and prevent physical damage to subtidal benthic invertebrates from mooring structures (Demers *et al.* 2013). Evidence for other interventions related to mooring are summarised under "Threat: Transportation and service corridors – Periodically move and relocate moorings", and "Provide additional moorings to reduce anchoring".

Demers M.C.A., Davis A.R. & Knott N.A. (2013) A comparison of the impact of 'seagrass-friendly'boat mooring systems on *Posidonia australis*. *Marine Environmental Research*, 83, 54–62.
Griffiths C.A., Langmead O.A., Readman J.A.J. & Tillin H.M. (2017) *Anchoring and Mooring Impacts in English and Welsh Marine Protected Areas: Reviewing sensitivity, activity, risk and management.* A report to Defra Impacts Evidence Group.

5.14. Periodically move and relocate moorings

- We found no studies that evaluated the effects of periodically moving and relocating moorings on subtidal benthic invertebrate 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

Moorings, permanent or temporary, provide an alternative to anchoring and allow for reduced anchoring in the area. Moorings can be provided for recreational vessels, as well as commercial vessels up to specific sizes. However, moorings also impact subtidal benthic invertebrates through chronic damage and physical disturbance to the seabed (Griffiths *et al*. 2017; Herbert *et al*. 2009). Periodically moving and relocating moorings within an area can allow the seabed and subtidal benthic invertebrate communities to naturally recover over time in the impacted zone (Griffiths *et al*. 2017; Herbert *et al*. 2009). Evidence for other interventions related to mooring are summarised under

"Threat: Transportation and service corridors – Use moorings which reduce or avoid contact with the seabed (eco-moorings)", and "Provide additional moorings to reduce anchoring".

Griffiths C.A., Langmead O.A., Readman J.A.J. & Tillin H.M. (2017) *Anchoring and Mooring Impacts in English and Welsh Marine Protected Areas: Reviewing sensitivity, activity, risk and management*. A report to Defra Impacts Evidence Group.

Herbert R.J., Crowe T.P., Bray S. & Sheader M. (2009) Disturbance of intertidal soft sediment assemblages caused by swinging boat moorings. *Hydrobiologia*, 625, 105–116.

5.15.Set limits on hull depth

- We found no studies that evaluated the effects of setting limits on hull depth on subtidal benthic invertebrate 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

Vessels can impact subtidal benthic invertebrates through disturbance to the seabed and water flow (Hammerstrom *et al*. 2007). Setting limits on hull depth of vessels (recreational or commercial) entering an area may potentially reduce disturbance to subtidal benthic invertebrates as the hull would be further from the seabed and less likely to result in direct physical damage from collision, grounding, or in changes in water flow or sediment transport (Whitfield *et al*. 2002).

Hammerstrom K.K., Kenworthy W.J., Whitfield P.E. & Merello M.F. (2007) Response and recovery dynamics of seagrasses *Thalassia testudinum* and *Syringodium filiforme* and macroalgae in experimental motor vessel disturbances. *Marine Ecology Progress Series* 345, 83–92.

Whitfield P., Kenworthy W., Hammerstrom K. & Fonseca M. (2002) The role of a hurricane in the expansion of disturbances initiated by motor vessels on seagrass banks. *Journal of Coastal Research*, 86–99.

5.16.Reduce ships/boats/vessels speed limits

- We found no studies that evaluated the effects of reducing ships/boats/vessels speed limits on subtidal benthic invertebrate 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

Vessel speed can induce waves and impact benthic invertebrates through disturbance to the seabed and changes in water flow (Gabel *et al.* 2012; Hammerstrom *et al*. 2007). Reducing vessel (recreational or commercial) speed limits in an area can potentially reduce the risk of changes in water flow or sediment transport, and benefit subtidal benthic invertebrates.

Gabel F., Garcia X.F., Schnauder I. & Pusch M.T. (2012) Effects of ship-induced waves on littoral benthic invertebrates. *Freshwater Biology*, 57, 2425–2435.

Hammerstrom K.K., Kenworthy W.J., Whitfield P.E. & Merello M.F. (2007) Response and recovery dynamics of seagrasses *Thalassia testudinum* and *Syringodium filiforme* and macroalgae in experimental motor vessel disturbances. *Marine Ecology Progress Series* 345, 83–92.

6. Threat: Biological resource use

Background

Biological resource use can have significant impacts on subtidal benthic invertebrates directly due to species extraction through harvest (reduced population of commercially targeted and non-targeted species [sometimes referred to as "bycatch"]) and indirectly through impacts on the seabed from fishing gear (modification and destruction of seabed habitats) (Collie *et al*. 2000; Lambert *et al.* 2014; Sciberras *et al.* 2018; Watling & Norse 1998). Interventions related to harvest restrictions or to the management of specific species within a Marine Protected Area are described in the "Habitat protection" chapter.

Please note that management interventions aimed at promoting the populations of commercial species are more closely related to harvest and fisheries management than conservation, and only the outcomes on the non-commercial species are included here. For instance, the conservation outcomes of interventions such as "Set commercial catch quotas" or "Restrict the use of a specific gear" for a specific commercial species (for instance cod) are not summarised for the commercially targeted species, but for any other subtidal benthic invertebrate species (i.e. unwanted catch species). We make one exception when the intervention is to stop the fishery altogether, for instance "ceasing or prohibiting harvest of conch" to conserve the conch population. In these instances, evidence is summarised in "Species management" (for instance "Species management – Cease or prohibit the harvest of conch"). Finally, interventions related to lost or abandoned fishing gear are described in "Threat: Pollution".

Additional threats and pressures related to biological resource use (mostly associated with fisheries management using spatial and temporal measures, catch quotas and effort control, or unwanted catch reduction) are described below.

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

Lambert G.I., Jennings S., Kaiser M.J., Davies T.W. & Hiddink J.G. (2014) Quantifying recovery rates and resilience of seabed habitats impacted by bottom fishing. *Journal of Applied Ecology*, 51, 1326–1336.

Sciberras M., Hiddink J.G., Jennings S., Szostek C.L., Hughes K.M., Kneafsey B., Clarke L.J., Ellis N., Rijnsdorp A.D., McConnaughey R.A., Hilborn R., Collie J.S., Pitcher C.R., Amoroso R.O., Parma A.M., Suuronen P. & Kaiser M.J. (2018) Response of benthic fauna to experimental bottom fishing: a global meta-analysis. *Fish & Fisheries*, 19, 698–715.

Watling L., & Norse E.A. (1998) Disturbance of the seabed by mobile fishing gear: a comparison to forest clearcutting. *Conservation Biology*, 12, 1180–1197.

Spatial and Temporal Management

6.1. Cease or prohibit all types of fishing

- **Five studies** examined the effects of ceasing or prohibiting all types of fishing on subtidal benthic invertebrate populations. All studies were in the North Sea[1–5] (Belgium, Germany, Netherlands, UK).

COMMUNITY RESPONSE (2 STUDIES)

- **Overall community composition (2 studies):** Two site comparison studies (one before-and-after) in the North Sea[1,3] found that areas closed to all fishing developed different overall invertebrate community compositions compared to fished areas.

- **Overall species richness/diversity (2 studies):** One of two site comparison studies (one before-and-after) in the North Sea[1,3] found that areas closed to all fishing did not develop different overall invertebrate species richness and diversity compared to fished areas after three years[3], but the other[1] found higher species richness in the closed areas after 20 years.

POPULATION RESPONSE (3 STUDIES)

- **Overall abundance (2 studies):** Two site comparison studies (one before-and-after) in the North Sea[3,4] found that areas closed to all fishing had similar overall invertebrate abundance and biomass compared to fished areas after three[3] and five[4] years.
- **Crustacean abundance (1 study):** One before-and-after, site comparison study in the North Sea[5] found that closing a site to all fishing led to similar numbers of lobster compared to a fished site after 20 months.
- **Crustacean condition (1 study):** One before-and-after, site comparison study in the North Sea[5] found that closing a site to all fishing led to larger sizes of lobster compared to a fished site after 20 months.

OTHER (1 STUDY)

- **Overall community energy flow (1 study):** One before-after, site comparison study in the North Sea[2] found that, during the 12–14 months after closing an area to all fishing, the invertebrate community structure (measured as energy flow) at sites within the closed area did not change, but that it increased in nearby fished sites.
- **Species energy flow (1 study):** One before-and-after, site comparison study in the North Sea[2] found that closing an area to all fishing for 12–14 months had mixed effects on species-level energy flow.

Background

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear coming into contact with the seabed (Collie *et al.* 2000). Ceasing or prohibiting all types of fishing in an area can remove the most direct pressures to subtidal benthic invertebrates, with the aim of enabling previously impacted populations to recover over time (Jack & Wing 2010). Evidence for related interventions is summarised under "Threat: Transportation and service corridors – Cease or prohibit shipping" and "Threat: Biological resource use – Establish temporary fisheries closure". When this intervention is undertaken within a protected area, evidence is summarised under "Habitat protection – Designate a Marine Protected Area and prohibit all types of fishing".

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

Jack L. & Wing S.R. (2010) Maintenance of old-growth size structure and fecundity of the red rock lobster Jasus edwardsii among marine protected areas in Fiordland, New Zealand. *Marine Ecology Progress Series,* 404, 161–172.

A site comparison study in 2004 in areas of soft sediment in the southern North Sea, Netherlands (1) found that an area closed to all fishing had different invertebrate community composition, and higher species richness, compared to fished areas, after approximately 20 years. Community data were presented as graphical analyses, and richness data were presented as diversity indices. A gas production platform was drilled approximately 20 years prior to the study and a 500 m zone closed to all trawling established around it. In April 2004, invertebrates were surveyed inside the closed area and in four sites (1 x 1 nm) outside (1.5 nm north, south, east and west of the exclusion zone). Samples were collected using a combination of dredge (6–10 tows/site; invertebrates >7 mm) and sediment cores (seven cores/site; invertebrates >1 mm) at 36–39 m depth. Invertebrates were identified and counted.

A before-after, site comparison study in 2003–2004 in areas of soft seabed sediment in the German Bight, southern North Sea, Germany (2) found that, during the 12–14 months after closing an area to all fishing, community structure (measured as energy flow) at sites within the closed area did not change, but increased in nearby open sites where fishing occurred. Before fishery closure, community energy flow was similar in the closed (80 kJ/m^2) and open sites (66 kJ/m^2). After 12–14 months, community energy flow in the closed sites was similar to before (69 kJ/m^2), but lower than at open sites where energy flow had increased over time (92 kJ/m^2). After 12 months, species-level energy flow was higher in closed areas compared to open areas for 10 of 70 species, and lower for 7 of 70 species. In July 2003 a pilot windfarm platform was constructed, and the surrounding area (500 m radius) closed to all shipping, and as such all fishing. Invertebrates were surveyed at 10 sites inside the windfarm area and 10–18 outside (9 km away) before construction (March–August), and 12–14 months after exclusion (July–October 2004). Invertebrates were collected using a sediment grab (0.1 m^2) and a beam trawl at 28 m depth. All were identified, counted, weighed, and their biomass converted to energy values (kilo Joule) using conversion factors. Energy flow was used to compare communities.

A before-and-after, site comparison study in 2008–2012 of multiple sites in an area of sandy seabed in the southern North Sea, 40–50 km off the coast of Belgium (3) found that over the three years after closing an area to all fishing, overall community composition changed over time both in closed sites and sites where fishing occurred, and were different overall, but total abundance, biomass, species richness and diversity did not change and remained similar across sites. Data and analyses of community compositions were not reported. Total invertebrate abundance did not change over time and remained similar at sites closed and open to fishing, both before (2008: closed 361 vs fished 436 individuals/m^2) and after (2011–2012: 369–1,027 vs 256–458) the closure. This was also true for total biomass (2008: 802 vs 1,656; 2011–2012: 514–5,733 vs 1,392–1,864 mg/m^2), species richness (2008: 10.3 vs 10.7; 2011–2012: 10.4–12.3 vs 10.3–14.7 species/sample), and diversity (reported as a diversity index). In 2009/2010 a windfarm was constructed, and an area of approximately 21 km^2 closed to all shipping, and as such fishing, established around the windfarm (500 m radius). Invertebrates were surveyed at 6–16 sites inside the windfarm area and 15–25 outside before construction in 2008, and after in 2011 and 2012 (always in September–October). Invertebrates >1 mm were collected using a sediment grab (0.1 m^2) at 15–40 m depth, identified, counted, and their dried biomass measured or estimated.

A site comparison study in 2011 of seven areas of soft seabed in the southern North Sea, Netherlands (4) found that overall, an area closed to all fishing had similar invertebrate abundance, biomass, species richness and diversity, compared to six adjacent fished areas, after five years. For each metric, not all data were shown. From core samples, all areas had similar invertebrate abundance (min. 1,096/m^2; (closed area); max. 1,778/m^2 (fished area)), biomass (min. 32 g/m^2 (closed area); max. 17 g/m^2 (fished area)), number of species (closed: 16; open: 13–20), and diversity (as diversity indices). From dredge samples, invertebrate abundance and species diversity were similar in the closed area and five of six fished areas, while all areas had similar biomass (min. 61 g/m^2 (closed area); max. 134 g/m^2 (closed area)) and number of species (closed: 20; fished: 15–21). An offshore wind farm was constructed in 2006, with a 500 m buffer zone approximately 25 km^2 around it closed to all shipping, and as such fishing. Invertebrates inside the closed area and at six nearby fished areas were surveyed in February 2011 using two methods. Shorter-lived infauna (>1 mm) were sampled using sediment core (0.078 m^2; 16 samples across the closed area; 8 samples/fished areas). Longer-lived infauna and epifauna (>7 mm) were sampled using a dredge (20 m^2; 14 samples across the closed area; 6 samples/fished areas). All invertebrates were identified, counted, and weighed (results are for dry weights).

A before-and-after, site comparison study in 2013–2015 of two rock and cobble sites off the northeast coast of the UK, North Sea (5) found that 20 months after closing a site to all fishing during wind farm construction, it had similar numbers but larger European lobster *Homarus gammarus* compared to a fished site. Total abundance increased in both sites and was similar between sites both before (closed: 63; fished: 74 lobsters/string) and after the closure (closed: 93; fished: 107). The proportion of large lobsters (>100 mm) increased in the closed site and was higher than in the fished site (data presented as size-frequency distributions). In addition, abundance of marketable lobsters (>87 mm) was similar between sites before closure (closed: 11; fished: 10 lobsters/string) but was higher in the closed site after 20 months (closed: 23; fished: 10). In 2014/2015 a 35 km^2 windfarm construction site approximately 10 km offshore was closed to all fishing for 20 months, until August 2015. Lobsters were surveyed at a site inside the windfarm area and a site outside (1 km north) in June–September 2013 and in June–September 2015. Each time at each site, 23–24 strings of 30 baited pots were deployed. Abundance (per string) and size of lobsters (carapace length) were recorded.

(1) Duineveld G.C.A., Bergman M.J.N. & Lavaleye M.S.S. (2007) Effects of an area closed to fisheries on the composition of the benthic fauna in the southern North Sea. *ICES Journal of Marine Science,* 64, 899–908.
(2) Dannheim J., Brey T., Schröder A., Mintenbeck K., Knust R. & Arntz W.E. (2014) Trophic look at soft-bottom communities — Short-term effects of trawling cessation on benthos. *Journal of Sea Research,* 85, 18–28.
(3) Coates D.A., Kapasakali D.A., Vincx M. & Vanaverbeke J. (2016) Short-term effects of fishery exclusion in offshore wind farms on macrofaunal communities in the Belgian part of the North Sea. *Fisheries Research,* 179, 131–138.
(4) Bergman M.J.N., Ubels S.M., Duineveld G.C.A. & Meesters E.W.G. (2015) Effects of a 5-year trawling ban on the local benthic community in a wind farm in the Dutch coastal zone. *ICES Journal of Marine Science,* 72, 962–972.
(5) Roach M., Cohen M., Forster R., Revill A.S., Johnson M. & Handling editor: Steven Degraer. (2018) The effects of temporary exclusion of activity due to wind farm construction on a lobster (*Homarus gammarus*) fishery suggests a potential management approach. *ICES Journal of Marine Science*, 75, 1416–1426.

6.2. Cease or prohibit commercial fishing

- **Three studies** examined the effects of ceasing or prohibiting commercial fishing on subtidal benthic invertebrate populations. Two studies were in the Tasman Sea[1,3] (New Zealand), the third on Gorges Bank in the North Atlantic Ocean[2] (USA).

COMMUNITY RESPONSE (2 STUDIES)

- **Overall community composition (1 study):** One site comparison study in the Tasman Sea[3] found that an area closed to commercial trawling and dredging for 28 years had different overall invertebrate communities than an area subject to commercial fishing.

- **Overall species richness/diversity (1 study):** One site comparison study on Georges Bank[2] found no difference in invertebrate species richness between an area closed to commercial fishing for 10 to 14 years and a fished area.

POPULATION RESPONSE (3 STUDIES)

- **Overall abundance (2 studies):** Two site comparison studies in the Tasman Sea[3] and on Georges Bank[2] found that areas prohibiting commercial fishing for 10 to 14 years[2] and 28 years[3] had greater overall invertebrate abundance compared to areas where commercial fishing occurred. One of the studies[3] also found higher biomass, while the other[2] found similar biomass in closed and fished areas.
- **Crustacean abundance (1 study):** One replicated, site comparison study in the Tasman Sea[1] found that in commercial fishing exclusion zones lobster abundance was not different to adjacent fished areas after up to two years.

OTHER (1 STUDY)

- **Overall community biological production (1 study):** One site comparison study in the Tasman Sea[3] found that an area closed to commercial trawling and dredging for 28 years had greater biological production from invertebrates than an area where commercial fishing occurred.

Background

Commercial fishing is one of the most wide-spread human impacts on the marine benthic environment (Thrush *et al.* 1998). It can impact subtidal benthic invertebrates through species removal or habitat damage and disturbance from fishing gear such towed trawls and dredges coming into contact with the seabed (Collie *et al*. 2000; Watling & Norse 1998). Ceasing or prohibiting commercial fishing in an area, but allowing other types of fishing (for instance recreational fishing, or for research purposes), can remove the most intense direct pressure to subtidal benthic invertebrates, and previously impacted populations are, in theory, able to recover over time (Hiddink *et al*. 2017; Kaiser *et al.* 2006). However, species and populations are still subjected to the effects of other fishing activities. When this intervention occurs within a protected area, evidence has been summarised under "Habitat protection – Designate a Marine Protected Area and prohibit commercial fishing". Evidence for related interventions is summarised under "Threat: Biological resource use – Cease or prohibit all towed (mobile) fishing gear", "Cease or prohibit bottom trawling" and "Cease or prohibit dredging".

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

Hiddink J.G., Jennings S., Sciberras M., Szostek C.L., Hughes K.M., Ellis N., Rijnsdorp A.D., McConnaughey R.A., Mazor T., Hilborn R. & Collie J.S. (2017) Global analysis of depletion and recovery of seabed biota after bottom trawling disturbance. *Proceedings of the National Academy of Sciences,* 114, 8301–8306.

Kaiser M.J., Clarke K.R., Hinz H., Austen M. C.V., Somerfield P.J. & Karakassis I. (2006) Global analysis of response and recovery of benthic biota to fishing. *Marine Ecology Progress Series*, 311, 1–14.

Thrush S.F., Hewitt J.E., Cummings V.J., Dayton P.K., Cryer M., Turner S.J., Funnell G.A., Budd R.G., Milburn C.J. & Wilkinson M.R. (1998) Disturbance of the marine benthic habitat by commercial fishing: impacts at the scale of the fishery. *Ecological Applications*, 8, 866–879.

A replicated, site comparison study in 2006–2007 of 12 rocky seabed sites in the Tasman Sea, Fiordland, New Zealand (1) found that a zone excluding commercial fishing did not have a higher abundance of red rock lobster *Jasus edwardsii* compared to adjacent fished areas, after up to two years. Lobster abundance was similar in the exclusion zone (2 individuals/250 m^2) and the fished areas (1 individual/250 m^2). In 2006 and 2007, divers surveyed eight sites within a commercial fishing exclusion zone set in 2005, and four fished sites (at 15 m depth). Red rock lobsters were counted along 50 x 5 m transects (1 transect/site in 2006, 3/site in 2007).

A site comparison study in 2004–2008 in two areas of gravelly and sandy seabed on Georges Bank, northwest Atlantic Ocean, USA (2) found that an area closed to certain commercial fishing had a higher biomass of invertebrates attached to the seabed (epifaunal), but not a higher total abundance or species richness, compared to a fished area, 10–14 years after closure. Epifaunal biomass was significantly higher in the closed area (33–109 g/L) compared to the fished area (26–57 g/L). Total epifauna abundance was similar in closed (6–15 individuals/L) and fished areas (6–10 individuals/L). The effect of closing commercial fishing on species richness varied with years, but overall across year species richness was similar in both areas (closed: 26–39; fished: 32–41 species). An area on Georges Bank was closed to all commercial fishing gear capable of retaining ground fish (trawls, scallop dredges, gill nets and hook gear) in December 1994. Annually in 2004–2008, one site in the closed area and one site in an adjacent fished area were surveyed at 45–55 m depth. Epifauna were collected using a dredge (2–3 samples/site/year; 6.4 mm mesh liner), identified, counted, and wet-weighed.

A site comparison study in spring 2008 of 48 sites in a soft seabed area in the Tasman Sea, New Zealand (3) found that sites within an area closed to commercial trawling and dredging for 28 years had different invertebrate communities, and higher invertebrate abundance, biomass and productivity than sites subject to intense fishing. Community data were presented as graphical analyses. Sites closed to fishing had greater invertebrate abundance (particularly large and small sizes, but not medium-size), and higher biomass and biological productivity, compared to fished sites (data presented as effect sizes). The larger, rarer individuals contributed the most to the biomass and productivity estimates within the closed sites. Separation Point exclusion zone was legally closed to commercial fishing and shellfish dredging in 1980. In 2008, sediments were collected from the western and southern edges of the exclusion zone, each with 12 samples on each side (24 samples inside and 24 outside the closed area in total) using a grab (0.07 m^2) at 20–30 m depth. Invertebrates (>0.5 mm) were extracted, identified and counted. Biomass and productivity were estimated using size-based conversion factors.

(1) Jack L. & Wing S.R. (2010) Maintenance of old-growth size structure and fecundity of the red rock lobster *Jasus edwardsii* among marine protected areas in Fiordland, New Zealand. *Marine Ecology Progress Series*, 404, 161–172.
(2) Smith B.E. Collie J.S. & Lengyel N.L. (2013) Effects of chronic bottom fishing on the benthic epifauna and diets of demersal fishes on northern Georges Bank. *Marine Ecology Progress Series*, 472, 199–217.
(3) Handley S.J., Willis T.J., Cole R.G., Bradley A., Cairney D.J., Brown S.N. & Carter M.E. (2014) The importance of benchmarking habitat structure and composition for understanding the extent of fishing impacts in soft sediment ecosystems. *Journal of Sea Research*, 86. 58–68.

6.3. Establish temporary fisheries closures

- **Six studies** examined the effects of establishing temporary fisheries closures on subtidal benthic invertebrates. One study was in the English Channel[1] (UK), one in the D'Entrecasteaux Channel[2] (Australia), one in the North Pacific Ocean[3] (USA), two in the Mozambique Channel[4a,b] (Madagascar), and one in the North Sea[5] (UK).

COMMUNITY RESPONSE (2 STUDIES)

- **Overall species richness/diversity (1 study):** One replicated, site comparison study in the English Channel[1] found that sites seasonally closed to towed-gear fishing did not have greater invertebrate species richness than sites where towed-fishing occurred year-round.
- **Mollusc community composition (1 study):** One replicated, before-and after study in the D'Entrecasteaux Channel[2] found that temporarily reopening an area previously closed to all fishing for 12 years only to recreational fishing led to changes in scallop species community composition over four fishing seasons.

POPULATION RESPONSE (1 STUDY)

- **Overall abundance (1 study):** One replicated, site comparison study in the English Channel[1] found that sites seasonally closed to towed-gear fishing did not have a greater invertebrate biomass than sites where towed-fishing occurred year-round.
- **Crustacean abundance (1 study):** One before-and-after, site comparison study in the North Sea[5] found that reopening a site to fishing following a temporary 20-month closure led to lower total abundance but similar marketable abundance of European lobsters compared to a continuously-fished site after a month.
- **Mollusc abundance (5 studies):** One replicated, site comparison study English Channel[1] found that sites seasonally closed to towed gear did not have higher abundance of great scallops than sites where towed-fishing occurred year-round. Two before-and after, site comparison studies (one replicated) in the Mozambique Channel[4a,b] found that temporarily closing an area to reef octopus fishing did not increase octopus abundance/biomass compared to before closure and to continuously fished areas. Two replicated, before-and after studies in the D'Entrecasteaux Channel[2] and the North Pacific Ocean[3] found that temporarily reopening an area previously closed to all fishing to recreational fishing only led to a decline in scallop abundance[2] after four fishing seasons and in red abalone[3] after three years.
- **Mollusc condition (3 studies):** One replicated, before-and after study in the North Pacific Ocean[3] found that temporarily reopening an area previously closed to fishing led to a decline in the size of red abalone after three years. Two before-and after, site comparison studies (one replicated) in the Mozambique Channel[4a,b] found that temporarily closing an area to reef octopus fishing increased the weight of octopus compared to before closure and to continuously fished areas, but one also found that this effect did not last once fishing resumed[4a].

Background

Establishing temporary fishery closures in an area can temporarily remove the most direct pressure from fishing to subtidal benthic invertebrates, and provide relief to impacted populations, which are, in theory, able to recover over time during the temporary closure (rotation) (Blyth *et al*. 2004; Hiddink *et al*. 2006; Rogers-Bennett *et al*. 2013). Temporary closures can include for instance: 1) seasonal closures, often done with the aim of protecting adults during the spawning season or to protect juveniles during times of recruitment or settlement, 2) rotational closures during which areas are alternately closed and opened to fishing following a specific timing, or 3) move-on rules whereby temporary closure of a fished area occurs when a catch or by-catch threshold is reached (Dunn *et al.* 2014). It is important to know how populations change once fisheries resume inside temporary closures, in order to understand whether the closures were effective, if the effects last, and if further measures and/or closures need to be implemented. Evidence for related interventions is summarised under "Threat: Biological resource use - Cease or prohibit all types of fishing" and "Cease or prohibit commercial fishing", and in "Habitat protection - Designate a Marine Protected Area and prohibit all types of fishing" and "Designate a Marine Protected Area and prohibit commercial fishing".

Blyth R.E., Kaiser M.J., Edwards-Jones G. & Hart P.J.B. (2004) Implications of a zoned fishery management system for marine benthic communities. *Journal of Applied Ecology*, 41, 951–961.
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.
Hiddink J.G., Hutton T., Jennings S. & Kaiser M. J. (2006) Predicting the effects of area closures and fishing effort restrictions on the production, biomass, and species richness of benthic invertebrate communities. *ICES Journal of Marine Science*, 63, 822–830.
Rogers-Bennett L., Hubbard K.E. & Juhasz C.I. (2013) Dramatic declines in red abalone populations after opening a "de facto" marine reserve to fishing: Testing temporal reserves. *Biological Conservation,* 157, 423–431.

A replicated, site comparison study in 2002 of five sites of mixed coarse seabed off the south Devon coast, English Channel, UK (1) found that sites seasonally closed to towed gear did not have greater invertebrate species richness or biomass, and did not have more great scallops *Pecten maximus* than sites where towed-fishing occurred year-round. Seasonal and year-round sites had similar average species richness (seasonal: 10–15 vs year-round: 8–10 species/tow), average biomass (1.5–2.4 vs 0.8–1.5 kg/tow), and average abundance of great scallops (1–11 vs 0–2 scallops/tow). In 1978 a zoned fishery management system was established in a 500 km^2 area, which included an area where towed-gear and static-gear rotated seasonally. In 2002, five sites were surveyed: two seasonally-towed and three towed year-round. Dredges were towed for 10 mins three times at each site (two standard dredges to collect great scallops >100 mm in length; one scientific dredge for other invertebrates). Species were identified and wet-weighed (individuals combined/species).

A replicated, before-and after study in 2005–2009 in a sandy seabed area in the D'Entrecasteaux Channel, southeastern Tasmania, Australia (2) found that temporarily reopening an area previously closed to all fishing to recreational fishing only led to changes in scallop species community composition over time and a 52% decline in overall scallop abundance after four fishing seasons. Community data were reported as statistical model result. Changes in scallop community composition over time was mostly due to changes in abundance of the scallop *Pecten fumatus*, which declined by 69% after four fishing seasons. In 2005, an area was reopened to scallop recreational fishing after 12 years of full fishing closure. Scallops (*Pecten fumatus, Equichlamys bifrons, Mimachlamys asperrimus*) were surveyed once before the first fishing season (in February 2006) and annually for four years after the fishing season (July–August 2006–2009). Twenty-four sites were surveyed in 2006–2007, and an additional 38 (total 62) in 2008–2009. Two divers identified and sized all scallops along a 200 m^2 transect at each site. No data prior to the closure are presented.

A replicated, before-and after study in 2004–2010 in a rocky seabed area in the North Pacific Ocean, northern California, USA (3) found that temporarily reopening an area previously closed to fishing led to a decline in abundance and size of red abalone *Haliotis rufescens* after three years. Abundance of abalone declined by 65% and was lower three years after reopening (0.45 abalone/m^2) compared to during closure (1.3 abalone/m^2). This was also true for the size of abalone (after reopening: 168 mm; during closure: 172 mm). Five months after closing the fisheries again, the abundance and size of abalone decreased further (abundance: 0.33/m^2; size: 166 mm). In July 2004, an area where abalone fishing had been prohibited was reopened to fishing. In May 2010, the area was designated as a marine protected area and closed again to fishing. Red abalone abundance and size were recorded along a total of 83 transects (60 m^2) in spring 2004 (prior to reopening fishing – 23 transects), September 2007 (during fishing – 33 transects), and September 2010 (five months after closing fishing again– 27 transects).

A before-and after, site comparison study in 2004–2005 in a coral reef area in the Velondriake Locally Managed Marine Area, southwestern Madagascar, Mozambique Channel (4a) found that temporarily closing an area to a reef octopus *Octopus cyanea* fishery did not lead to a significant increase in the weight of octopus, and did not increase their abundance, compared to before closure and to continuously fished areas. Across fishing events during the first spring tide following the temporary closure, the average weight of caught octopus was 64% higher than before closure (before: 719 g; after: 1,120 g) but this was not statistically significant. In addition, the percentage of caught octopus over 2 kg increased from 8% to 20%. However, this increase in weight was not observed across the second spring tide (data not shown). Abundance (as biomass of octopus caught) did not change before and after closure in either temporarily closed sites or continuously fished sites and was similar at all sites (closed before: 3, closed after: 2–3.5; fished before: 2.7, fished after: 2–2.5 kg/fisher/day). An octopus fishery was closed at one site between November 2004 and June 2005 by means of a *Dina* (a traditional local law). Fishery data, including octopus weight, catch/unit effort and location of landing, were collected on a regular basis across nine nearby villages from September 2004 (before closure) and until at least two spring tides after reopening. These data included the closed site and 14 continuously fished sites (where spear fishing was the only practice).

A replicated, before-and after, site comparison study in 2005–2006 in a coral reef area in the Velondriake Locally Managed Marine Area, southwestern Madagascar, Mozambique Channel (4b) found that temporarily closing areas to a reef octopus *Octopus cyanea* fishery led to an increase in the weight of octopus, but not abundance, compared to before closure and to continuously fished areas. Across fishing events during the first spring tide following the temporary closure, the average weight of caught octopus was 160% higher at one of the three sites than before closure (before: 436 g; after: 1,136 g). However, this effect was not observed across the second spring tide (data not shown). At the two other closed sites, octopus weight was similar before closure (893 g and 997 g) and directly after reopening (889 g and 988 g). However, on reopening, octopus at all closed sites were 21–56% bigger (by weight; 889–1,165 g) than at fished sites (737 g). Following closure, abundance (as biomass of octopus caught) had increased by 88–146% in the closed sites (before: 1.3–1.6; after: 3–3.2 kg/fisher/day), while abundance did not change at fished sites and was lower (before: 2.4; after: 2.8 kg/fisher/day). This effect was also observed in the following tides. A state-enacted closure of octopus fishery was set across southwest Madagascar between early December 2005 and end of January 2006. This closure was extended at three sites and set between November 2005 and April 2006 by means of a *Dina* (a traditional local law). Fishery data, including octopus weight, catch/unit effort, and location of landing, were collected on a regular basis across nine nearby villages from September 2004 (before closure) until September 2006 (at least two spring tides after reopening). These data included the closed sites and 14 continuously fished sites (where spear fishing wass the only practice).

A before-and-after, site comparison study in 2013–2015 of two rock and cobble sites off the Holderness coast, northeast UK, North Sea (5) found that reopening a site to fishing following a temporary 20-month closure during wind farm construction led to lower total abundance but similar marketable abundance of European lobsters *Homarus gammarus* a month after fishing resumed compared to a continuously-fished site. Total abundance was similar at both sites after 20 months of closure and before reopening (reopened: 113; fished: 107 lobsters) but reduced at the reopened site after a month (72), with no change at the fished site (108). Before reopening, the abundance of marketable lobsters (>87 mm) was higher at the reopened site (37) compared to the fished site (12) but decreased at both sites to similar levels following reopening (reopened: 9; fished: 8). In 2014–2015 a 35 km^2 windfarm was constructed approximately 10 km offshore. The area was closed to all fishing for 20 months during construction, until August 2015. Lobsters were surveyed at a site inside the windfarm area and a site outside (1 km north) in June 2015 (before reopening) and in September 2015 (after reopening). During each survey, 11–13 strings of 30 baited pots were deployed at each site. Abundance (per string) and size of lobsters (carapace length) were recorded.

(1) Blyth R.E., Kaiser M.J., Edwards-Jones G. & Hart P.J.B. (2004) Implications of a zoned fishery management system for marine benthic communities. *Journal of Applied Ecology*, 41, 951–961.
(2) Tracey S.R. & Lyle J.M. (2011) Linking scallop distribution and abundance with fisher behaviour: implication for management to avoid repeated stock collapse in a recreational fishery. *Fisheries Management and Ecology*, 18, 221–232.
(3) Rogers-Bennett L., Hubbard K.E. & Juhasz C.I. (2013) Dramatic declines in red abalone populations after opening a "de facto" marine reserve to fishing: Testing temporal reserves. *Biological Conservation,* 157, 423–431.
(4a-b) Benbow S., Humber F., Oliver T.A., Oleson K.L.L., Raberinar, D., Nado, M., Ratsimbazaf, H. & Harris A. (2014) Lessons learnt from experimental temporary octopus fishing closures in south-west Madagascar: benefits of concurrent closures. *African Journal of Marine Science*, 36, 31–37.
(5) Roach M., Cohen M., Forster R., Revill A. S., Johnson M. & Handling editor: Steven Degraer. (2018) The effects of temporary exclusion of activity due to wind farm construction on a lobster (*Homarus gammarus*) fishery suggests a potential management approach. *ICES Journal of Marine Science*, 75, 1416–1426.

Mobile fishing gear

6.4. Cease or prohibit bottom trawling

- **Four studies** examined the effects of ceasing or prohibiting bottom trawling on subtidal benthic invertebrate populations. Two studies were in the Bering Sea[1,3] (USA), one in the North Sea[2], and one in the Mediterranean Sea[4] (Italy).

COMMUNITY RESPONSE (3 STUDIES)

- **Overall community composition (2 studies):** Two site comparison studies (one before-and-after, one replicated) in the North Sea[2] and the Mediterranean Sea[4] found that in areas prohibiting trawling for either 15 or 20 years, overall invertebrate community composition was different to that of trawled areas.
- **Overall species richness/diversity (3 studies):** Two of three site comparison studies (one paired, one before-and-after, one replicated) in the Bering Sea[1], the North Sea[2], and the Mediterranean Sea[4] found that invertebrate diversity was higher in sites closed to trawling compared to trawled sites after either 37 or 15 years[1,2], but the other[4] found no differences after 20 years.

POPULATION RESPONSE (3 STUDIES)

- **Overall abundance (2 studies):** One of two site comparison studies (one paired, one replicated) in the Bering Sea[1] and the Mediterranean Sea[4] found that total invertebrate abundance was higher in sites closed to trawling compared to trawled sites after 37 years[1], but the other[4] found no differences after 20 years. Both found no differences in total invertebrate biomass.

- **Unwanted catch overall abundance (1 study):** One replicated, before-and-after, site comparison study in the Bering Sea[3] found that during the three years after closing areas to all bottom trawling, unwanted catch of crabs appeared to have decreased, while no changes appeared to have occurred in nearby trawled areas.

Background

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear coming into contact with the seabed (Collie *et al*. 2000). Mobile fishing gear such as bottom trawls are known to be particularly damaging as they are dragged along the seabed. Their use can be stopped or prohibited within an area and only static gears (such as lobster or crab pots/traps) allowed. Ceasing or prohibiting bottom trawling can remove this direct pressure to subtidal benthic invertebrates and potentially allow them to recolonise and recover naturally over time (Hiddink *et al*. 2017). When this intervention occurs due to the closure of an area to shipping, evidence has been summarised under "Threat: Transportation and service corridors – "Cease or prohibit shipping". When it is in combination with ceasing or prohibiting dredging, but without separating the effects of the two gear, evidence has been summarised under "Threat: Biological resource use – Cease or prohibit all towed (mobile) fishing gear". When this intervention occurs within a protected area, evidence has been summarised under "Habitat protection – Designate a Marine Protected Area and prohibit bottom trawling".

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

Hiddink J.G., Jennings S., Sciberras M., Szostek C.L., Hughes K.M., Ellis N., Rijnsdorp A.D., McConnaughey R.A., Mazor T., Hilborn R. & Collie J.S. (2017) Global analysis of depletion and recovery of seabed biota after bottom trawling disturbance. *Proceedings of the National Academy of Sciences,* 114, 8301–8306.

A paired site comparison study in 1996 of 84 sites of sandy seabed in the eastern Bering Sea, USA (1) found that 'macro'-invertebrate (size unspecified) species diversity and abundance were higher in sites closed to trawling for 37 years, compared to trawled sites, but there was no difference in biomass. Overall across paired sites, species diversity was higher in sites closed to trawling compared to those trawled (reported as a diversity index). Of the 42 invertebrate taxa recorded, 27 appeared more abundant in the closed sites compared to the trawled sites (not statistically tested). In particular, abundances of sponge (*Porifera*), anemones (*Actinaria*) and *Neptunea* snails (gastropods) were significantly higher in the closed sites (data not shown). Invertebrate biomass was similar in sites closed to trawling (1.6 kg/ha) and trawled sites (1.6 kg/ha). Trawling was prohibited in an area in 1959. Macro-invertebrates were surveyed at 84 sampling sites (44–55 m depth) along the boundary of the closed area (42 pairs; one site on either side of the boundary, 1 nm apart) using an otter trawl (3.8 cm liner at the codend). Macro-invertebrates were sorted into groups, counted and weighed.

A site comparison study in 2004 in areas of soft sediment in the southern North Sea, Netherlands (2) found that an area closed to bottom trawling had different invertebrate community composition, and higher species richness, compared to areas where trawling occurred, after approximately 20 years. Community data were presented as graphical analyses, and richness data were presented as a diversity index. A gas production platform was drilled approximately 20 years prior to the study and a 500 m zone closed to all trawling, established around it. In April 2004, invertebrates were surveyed inside the closed area and in four sites (1 x 1 nm) outside (1.5 nm north, south, east and west of the exclusion zone). Samples were collected using a combination of dredge (6–10 tows/site; invertebrates >7 mm) and sediment cores (seven cores/site; invertebrates >1 mm) at 36–39 m depth. Invertebrates were identified and counted.

A replicated, before-and-after, site comparison study in 1992–1997 in the eastern Bering Sea, USA (3) found that during the three years after closing areas to all bottom trawling, unwanted catch of red king crab *Paralithodes camtschaticus* appeared to have decreased, while no changes appeared to have occurred in nearby trawled areas (results not tested for statistical significance). In the closed areas, average unwanted crab catch tended to be lower after the closure (2–4 crabs/hour) compared to before (6–17). In addition, the proportion of hauls without crabs tended to be higher after the closure (after: 91–95%) compared to before (71–86%). In the continuously trawled areas, unwanted crab catch was similar before (2–8 crabs/hour) and after (2–4 crabs/hour) the closure. Two areas were closed to all bottom trawling in 1995. Unwanted catch data inside the closed areas and in nearby trawled areas (number and location unspecified) between January 1992 and March 1997 were obtained from the North Pacific Groundfish Observer Program (approximately 4,500 observations).

A replicated, site comparison study in 2005 in four gulfs of muddy seabed in the Mediterranean Sea, off the northern coast of Sicily, Italy (4) found that, 15 years after prohibiting trawling, overall invertebrate community composition, but not total invertebrate abundance, biomass, or diversity, was different to that of trawled gulfs. Invertebrate communities were different between non-trawled and trawled gulfs (community data presented as graphical analyses), with amphipods reported to dominate non-trawled gulfs, while polychaete worms reported to dominate trawled gulfs. There were no statistical differences between gulfs in total abundance (non-trawled: 683–872; trawled: 448–633 individuals/m^2), total biomass (non-trawled: 751–927; trawled: 1,000–1,080 g/m^2) and diversity (as a diversity index). Two gulfs (200 and 240 km^2) were closed to trawling in 1990 (artisanal fishing with static gears and small purse seines allowed). In May–June 2005, sediment samples were collected in the two closed gulfs and two fished gulfs (18 samples/gulf) using a grab (0.4 m^2; 3 grabs/sample) at 40–80 m depth. Invertebrates >0.5 mm were identified to family level and dry-weighed.

(1) McConnaughey R.A., Mier K.L. & Dew C.B. (2000) An examination of chronic trawling effects on soft-bottom benthos of the eastern Bering Sea. *ICES Journal of Marine Science,* 57, 1377–1388.
(2) Duineveld G.C.A., Bergman M.J.N. & Lavaleye M.S.S. (2007) Effects of an area closed to fisheries on the composition of the benthic fauna in the southern North Sea. *ICES Journal of Marine Science,* 64, 899–908.
(3) Abbott J.K. & Haynie A.C. (2012) What are we protecting? Fisher behavior and the unintended consequences of spatial closures as a fishery management tool. *Ecological Applications*, 22, 762–777.
(4) Romano C., Fanelli E., D'Anna G., Pipitone C., Vizzini S., Mazzola A. & Badalamenti F. (2016) Spatial variability of soft-bottom macrobenthic communities in northern Sicily (Western Mediterranean): Contrasting trawled vs. untrawled areas. *Marine Environmental Research,* 122, 113–125.

6.5. Cease or prohibit midwater/semi-pelagic trawling

- We found no studies that evaluated the effects of ceasing or prohibiting midwater/semi-pelagic trawling on subtidal benthic invertebrate 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

Many populations of subtidal benthic invertebrate species have declined or been depleted due to the multiple threats they are exposed to, including overharvesting (Hobday *et al*. 2000) and unintentional physical damage or catching during other fishing operations (Collie *et al*. 2000). Midwater/semi-pelagic trawling, which tows nets at depths higher in the water column (shallower) than bottom trawling, has in theory less impacts on the seabed and benthic invertebrates, as the gear should not come into contact with them. However, midwater trawls sometimes do come into contact with the seabed or with benthic invertebrates, particularly in areas with uneven topography and geological features, such as seamounts (Clark & Koslow 2007; He & Winder 2010). Therefore, ceasing or prohibiting midwater/pelagic trawling in an area can reduce or remove this pressure, and potentially benefit subtidal benthic invertebrate populations. Evidence for related interventions is summarised under "Threat: Biological resource use – Use a semi-pelagic trawl instead of demersal trawl", and additional evidence related to ghost fishing from abandoned or lost gear is summarised under "Threat: Pollution - Use biodegradable panels in fishing pots" and "Recover lost fishing gear".

Clark M.R. & Koslow J.A. (2007) Impacts of fisheries on seamounts*. Seamounts: Ecology, Fisheries, and Conservation*, 12, 413–441.

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology,* 69, 785–798.

He P. & Winger P.D. (2010) Effect of Trawling on the Seabed and Mitigation Measures to Reduce Impact. Pages 295–314 in: P. He (Ed.) *Behavior of Marine Fishes*.

Hobday A.J., Tegner M.J. & Haaker P.L. (2000) Over-exploitation of a broadcast spawning marine invertebrate: decline of the white abalone. *Reviews in Fish Biology and Fisheries*, 10, 493–514.

6.6. Cease or prohibit dredging

- **Four studies** examined the effects of ceasing or prohibiting dredging on subtidal benthic invertebrate populations. One study was in the North Atlantic Ocean[1] (Portugal), one in the South Atlantic Ocean[2] (Argentina), one in the English Channel[3] and one in the Irish Sea[4] (UK).

COMMUNITY RESPONSE (3 STUDIES)

- **Overall community composition (3 studies):** One of three site comparison studies (one replicated, one before-and-after) in Atlantic Ocean[1,2] and the Irish Sea[4] found that after ceasing dredging, overall invertebrate community composition was different to that in dredged areas[1]. The other two[2,4] found that communities remained similar in dredged and non-dredged areas.
- **Overall richness/diversity (3 studies):** One of three site comparison studies (one replicated, one before-and-after) in Atlantic Ocean[1,2] and the Irish Sea[4] found that after ceasing dredging, large (macro-) invertebrate diversity was higher but small (meio-) invertebrate diversity was lower compared to dredged areas[1]. The other two[2,4] found that overall diversity remained similar in dredged and non-dredged areas.

POPULATION RESPONSE (3 STUDIES)

- **Overall abundance (3 studies):** One of three site comparison studies (one replicated, one before-and-after) in Atlantic Ocean[1,2] and the Irish Sea[4] found that four years after ceasing dredging, large (macro-) and small (meio-) invertebrate abundance and/or biomass appeared higher to that in dredged areas[1]. The other two[2,4] found that abundance and/or biomass remained similar in dredged and non-dredged areas after either two[4] or six[2] years.

- **Tunicate abundance (1 study):** One replicated, site comparison study in the English Channel[3] found that a year after ceasing dredging in three areas, abundance of ascidians/sea squirts (tunicates) was similar to that in dredged areas.
- **Bryozoan abundance (1 study):** One replicated, site comparison study in the English Channel[3] found that a year after ceasing dredging in three areas, abundance of bryozoan was higher than in dredged areas.
- **Crustacean abundance (1 study):** One replicated, site comparison study in the English Channel[3] found that a year after ceasing dredging in three areas, abundance of spider crabs was higher than in dredged areas, but abundance of edible crab was similar.
- **Cnidarian abundance (1 study):** One replicated, site comparison study in the English Channel[3] found that a year after ceasing dredging in three areas, abundance of sea fans was higher than in dredged areas.
- **Sponge abundance (1 study):** One replicated, site comparison study in the English Channel[3] found that a year after ceasing dredging in three areas, abundance of sponges was higher than in dredged areas.

Background

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear coming into contact with the seabed (Collie *et al*. 2000). Mobile fishing gear such as towed dredges, for instance used in the harvest of bivalves such as clams and scallops, involves towing a heavy steel frame along the seabed, and are known to be particularly damaging to benthic biota. Ceasing or prohibiting dredges in an area, for instance through bylaws or voluntary agreements (Blyth *et al.* 2002; Bull 1989; Schejter *et al.* 2008), can remove this direct pressure to subtidal benthic invertebrates and allow them to potentially recolonise and recover naturally over time (Blyth *et al*. 2004). Recreational and artisanal bivalve fishing may cause less impact compared to dredging due to the smaller scale of the operations and the harvesting methods used (for instance hand-harvest). Evidence for related interventions is summarised under "Threat: Biological resource use – Cease or prohibit all towed (mobile) fishing gear". When this intervention occurs within a protected area, evidence has been summarised under "Habitat protection – Designate a Marine Protected Area and prohibit dredging" and "Designate a Marine Protected Area and prohibit the harvest of scallop".

Blyth R.E., Kaiser M.J., Edwards-Jones G. & Hart P.J.B. (2004) Implications of a zoned fishery management system for marine benthic communities. *Journal of Applied Ecology*, 41, 951-961.

Blyth R.E., Kaiser M. J., Edwards-Jones G. & Hart P.J.B. (2002) Voluntary management in an inshore fishery has conservation benefits. *Environmental Conservation*, 29, 493–508.

Bull M.F. (1989) *The New Zealand scallop fishery: a brief review of the fishery and its management*. Edited by: MLC Dredge, WF Zacharin and LM Joli, 42.

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

Schejter, L., Bremec, C.S. & Hernández, D. (2008) Comparison between disturbed and undisturbed areas of the Patagonian scallop (*Zygochlamys patagonica*) fishing ground "Reclutas" in the Argentine Sea. *Journal of Sea Research*, 60, 193–200.

A replicated, site comparison study in 1999 of six sandy seabed sites off the Algarve coast, North Atlantic Ocean, southwestern Portugal (1) found that sites closed to dredging had different invertebrate community composition, higher macro-invertebrate (>1 mm) diversity, but lower meio-invertebrate (150 μm–1mm) diversity after four years, than sites where dredging continued. Communities in the closed and fished areas were 88% dissimilar (data presented as statistical model result). Macro-invertebrate diversity was higher, but meio-invertebrate diversity was lower, inside the closed area compared to the fished areas (reported as diversity indices). Macro-invertebrate abundance averaged 12 individuals/m^2 in the closed area, and 4 individuals/m^2 in the fished area. Macro-invertebrate biomass averaged 0.61 g/m^2 in the closed area, and 0.65 g/m^2 in the fished area. Meio-invertebrate abundance averaged 49 individuals/m^2 in the closed area, and 42 individuals/m^2 in the fished area. Meio-invertebrate biomass averaged 5 g/m^2 in the closed area, and 0.1 g/m^2 in the fished area. Abundance and biomass data were not statistically tested. In 1995, an area was closed to dredge fishing (whether other fishing activities continued is unclear). Invertebrates were surveyed at three 50 x 50 m sites in the closed area and three in a nearby area where dredging continued (7–9 m depth) using quadrats and cores. Macro- and meio-invertebrates were identified, counted, and dry-weighed.

A site comparison study in 1998–2002 in two areas of soft seabed in the South Atlantic Ocean, Argentina (2) found that an area prohibiting the commercial dredging of scallops for six years did not have different invertebrate community composition, species richness, or biomass, compared to adjacent fished areas. Community data were presented as graphical analyses. Species richness was similar in closed areas (11–24 species groups/site) and fished area (6–25 species groups/site) throughout the study. Six years after closure, biomass of invertebrates was similar in the closed (2–13 kg/100 m^2) and fished areas (2–16 kg/100 m^2). The area was closed to commercial dredging of scallops in 1996. Samples were collected at 100 m depth once a year between 1998 and 2002 using a dredge (which does not catch scallops; 10 mm mesh) at 23 sites in the closed area and at 71 adjacent sites outside. Invertebrates were identified to species level when possible, counted and weighed. Information was updated using an erratum (Schejter *et al*. 2009).

Schejter L., Bremec C.S. & Hernández D. (2009) Erratum to "Comparison between disturbed and undisturbed areas of the Patagonian scallop (*Zygochlamys patagonica*) fishing ground "Reclutas" in the Argentine Sea" [J. Sea Research 60/3 (2008) 193]. *Journal of Sea Research* 61, 275.

A replicated, site comparison study in 2007 in six areas of rocky seabed in Lyme Bay, English Channel, UK (3) found that closing areas to scallop dredging had mixed effects on the abundance of invertebrates depending on species, after a year. Abundances were higher in the closed areas, compared to areas that remained dredged, for pink sea fans *Eunicella verrucosa* (closed: 58 vs dredged: 15 individuals/100 m^2), bryozoans *Pentapora fascialis* (27 vs 9 individuals/100 m^2), sponges *Axinella dissimilis* (5.0 vs 1.4 individuals/100 m^2), and spider crabs *Maja squinado* (1.2 vs 0.3 individuals/100 m^2). In contrast, there was no difference in abundance between areas for tunicates (ascidian/sea squirt) *Phallusia mammillata* (6 vs 12 individuals/100 m^2), or edible crabs *Cancer pagurus* (1 vs 1 individuals/100 m^2). In March and August 2007, six areas within the bay were sampled: three voluntarily closed to scallop dredging since September 2006 (but where static gear fisheries occurred) and three that remained open. Samples were taken using a video camera (10 recordings/area) towed for approximately 10 minutes in a straight line. Abundances of six species were recorded from the videos.

A before-and-after, site comparison study 2009–2011 in two areas of sandy, pebbly and gravelly seabed in Cardigan Bay, Irish Sea, Wales, UK (4) found that in an area prohibiting commercial scallop dredging year-round, sessile invertebrate community composition, diversity, species richness, and abundance were similar to that of an adjacent seasonally dredged area, after two years. Invertebrate community composition (presented as graphical analyses), diversity (presented as a diversity index), species richness, and abundance, were similar between closed and fished areas both before (richness: closed 7 vs fished 4 species/tow; abundance: 3 vs 3 individuals/m^2) and two years after closure (richness: 15 vs 13 species/tow; abundance: 23 vs 7 individuals/m^2). Richness, diversity, abundance and assemblage composition changed in a similar manner over time in the closed and fished areas. Two areas of Cardigan Bay were assessed: one permanently closed to scallop dredging in March 2010, another seasonally closed to scallop dredging (May to October). Surveys were conducted before closure (December 2009) and three times after (June 2010 to April 2011). During each survey, a camera was towed behind a boat at 30 m depth for 300 m at six sites/area. More than 40 images/camera tow (covering a 0.13 m^2 area of seabed) were analysed, and sessile invertebrates were identified and counted.

(1) Chícharo L., Chícharo A., Gaspar M., Alves F. & Regala J. (2002) Ecological characterization of dredged and non-dredged bivalve fishing areas off south Portugal. *Journal of the Marine Biological Association of the United Kingdom*, 82, 41–50.
(2) Schejter L., Bremec C.S. & Hernández D. (2008) Comparison between disturbed and undisturbed areas of the Patagonian scallop (*Zygochlamys patagonica*) fishing ground "Reclutas" in the Argentine Sea. *Journal of Sea Research*, 60, 193–200.
(3) Hinz H., Tarrant D., Ridgeway A., Kaiser M.J. & Hiddink J.G. (2011) Effects of scallop dredging on temperate reef fauna. *Marine Ecology Progress Series*, 432, 91–102.
(4) Sciberras M., Hinz H., Bennell J.D., Jenkins S.R., Hawkins S.J. & Kaiser M.J. (2013) Benthic community response to a scallop dredging closure within a dynamic seabed habitat. *Marine Ecology Progress Series,* 480, 83–98.

6.7. Cease or prohibit all towed (mobile) fishing gear

- **Eight studies** examined the effects of ceasing or prohibiting all towed fishing gear on subtidal benthic invertebrate populations. One study was in the Limfjord[1] (Denmark), two in the English Channel[2,4] (UK), three in Georges Bank in the North Atlantic Ocean[3,7,8] (USA and Canada), one in the Ria Formosa lagoon[6] (Portugal), and one in the Irish Sea[5] (Isle of Man).

COMMUNITY RESPONSE (4 STUDIES)

- **Overall community composition (3 studies):** Two of three replicated, site comparison studies in the Limfjord[1] and the English Channel[2,4], found that areas excluding towed fishing gear for either an unspecified amount of time[2] or two to 23 years[4] had different overall invertebrate community composition compared to areas where towed-fishing occurred[2,4] and one[1] found that ceasing towed-gear fishing for nine years had mixed effects.
- **Overall species richness/diversity (3 studies):** Two replicated, site comparison studies in the English Channel[2,4] reported that areas excluding towed fishing gear for either an unspecified amount of time[2] or two to 23 years[4] had different[2] or greater[4] invertebrate species richness and diversity to areas where towed-fishing occurred. One site comparison study in Georges Bank[8] found no difference in invertebrate species richness between an area closed to mobile fishing gear for 10 to 14 years and a fished area.

POPULATION RESPONSE (3 STUDIES)

- **Overall abundance (3 studies):** Two site comparison studies (one replicated) in the English Channel[4] and Georges Bank[8] found that sites excluding towed gear for either two to 23 years[4] or 10 to 14 years[8] had greater overall invertebrate biomass compared to sites where towed-gear fishing occurred, but one[8] also found that abundance was similar in both areas. One replicated, controlled, before-and-after study in the Ria Formosa lagoon[6] found that ceasing towed gear for 10 months led to increases in the cover of mobile but not sessile invertebrates.

- **Mollusc abundance (2 studies):** Two site comparison studies (one replicated) in the Irish Sea[5] and the English Channel[4] found that areas closed to towed fishing gear for either two to 23 years[4] or 14 years[5] had more scallops compared to adjacent fished areas.
- **Mollusc condition (1 study):** One site comparison study the Irish Sea[5] found that an area closed to towed fishing gear for 14 years had higher proportions of older and larger scallops compared to an adjacent fished area.
- **Starfish abundance (1 study):** One replicated, site comparison study in Georges Bank[7] found more starfish in areas closed to towed fishing gear for five to nine years compared to adjacent fished areas.
- **Starfish condition (1 study):** One replicated, site comparison study in Georges Bank[7] found that starfish arm length was similar in areas closed to towed fishing gear for five to nine years and adjacent fished areas.

OTHER (1 STUDY)

- **Overall community biological production (1 study):** One before-and-after, site comparison study in Georges Bank[3] found an increase in the biological production from invertebrate in sites closed to towed fishing gear for approximately five years compared to adjacent fished sites.

Background

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear coming into contact with the seabed (Collie *et al.* 2000). Bottom trawls and dredges are mobile fishing gears towed behind a vessel and are known to be particularly damaging as they are dragged onto the seabed (mid-water trawls can also sometimes accidentally come into contact with the seabed). Ceasing or prohibiting all towed gears in an area, for instance through bylaws or voluntary agreements (Blyth *et al.* 2002), can remove their direct pressure on subtidal benthic invertebrates and potentially allow them to recolonise and recover naturally over time (Blyth *et al.* 2004). Evidence related to ceasing only trawls or only dredges are summarised under "Threat: Biological resource use – Cease or prohibit bottom trawling" and "Cease or prohibit dredging", respectively. When the cessation of towed fishing gear occurs in the context of a marine protected area, the evidence has been summarised under "Habitat protection – Designate a Marine Protected Area and prohibit all towed (mobile) fishing gear".

Blyth R.E., Kaiser M.J., Edwards-Jones G. & Hart P.J.B. (2004) Implications of a zoned fishery management system for marine benthic communities. *Journal of Applied Ecology*, 41, 951–961.

Blyth R.E., Kaiser M.J., Edwards-Jones G. & Hart P.J.B. (2002) Voluntary management in an inshore fishery has conservation benefits. *Environmental Conservation*, 29, 493–508.

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

A replicated, site comparison study in 1997 of 17 sites in the Limfjord, northern Denmark (1) found that ceasing towed gear fishing in an area for nine years had mixed effects on invertebrate community composition. Sites in the northern part of the area closed to towed gear had different invertebrate composition to adjacent northern fished sites, but sites in the southern part of the closed area had similar assemblages to adjacent southern fished sites (community data were presented as graphical analyses and statistical model results). Within the closed area, northern sites also had different composition to southern sites. Authors suggest towed gears might not have been the cause of the observed changes in invertebrate and fish compositions prior to the closure. A 40 km^2 area was closed to towed gears (static gears allowed) in 1988 following changes in invertebrate and fish assemblages. In September 1997, divers identified and counted sessile invertebrates at 17 sites (ten 0.24 m^2 quadrats/site) across four areas: northern fished area (four sites), northern closed area (five sites), southern closed area (four sites), and southern fished area (four sites).

A replicated, site comparison study (year not stated) in eight areas of mixed sediment off the south Devon coast, English Channel, UK (2) found that areas excluding towed fishing gear (for an unspecified amount of time) had different species richness, diversity and overall invertebrate community composition compared to areas where towed-fishing occurred either seasonally or year-round. Species richness and diversity data were not presented. Community composition in areas closed to towed gears was reported to be dominated by higher biomass and organisms that increased habitat complexity (community data were presented as graphical analyses). In areas where towed-fishing occurred, the community was reported to be dominated by smaller bodied fauna and scavenging taxa. In 1978 a zoned fishery management system was established in a 500 km^2 area, which included static-gear-only areas. Eight areas were surveyed (year of study unspecified) at 15–70 m depth: three non-towed (static only), two seasonally-towed (six months/year), and three towed year-round. Invertebrates were sampled at nine stations/area. Invertebrates were sampled with a beam trawl and a dredge, identified, counted and weighed.

A before-and-after, site comparison study in 1994–2000 of four sites of sandy and gravelly seabed on Georges Bank, North Atlantic Ocean, USA and Canada (3) found an increase in invertebrate biological production in shallow and deep sites closed to towed fishing gear compared to adjacent fished sites approximately five years after closure. Biological production (a measure of biomass regeneration over time) from invertebrates at shallow (45–62 m) sites closed to fishing increased following closure (before: 17; after: 215 kcal/m^2/year), and was higher than at shallow fished sites where production did not vary over time (before: 32; after: 57 kcal/m^2/year). Production at deep (80–90 m) sites closed to fishing also increased following closure (before: 174; after: 256 kcal/m^2/year), and was higher than at deep fished sites where production did not vary over time (before: 52; after: 30 kcal/m^2/year). In January 1995, a combined area of approximately 10,000 km^2 of Georges Bank was closed to all bottom towed fishing gear. Invertebrates (>5 mm) were sampled with a dredge (6.4 mm mesh) at four sites across the two depth ranges ('shallow' and 'deep'). Shallower sites are subject to more intense and regular fishing. At each depth, one closed and one fished site were sampled. Animals were identified, counted and weighed. Individuals from the 20 most abundant species were measured. Biological production was estimated from a combination of biomass and length-frequency distribution data.

A replicated, site comparison study in 2002 of seven sites of mixed coarse seabed off the south Devon coast, English Channel, UK (4) found that sites excluding towed gear, for either two or 24 years, had greater invertebrate species richness and biomass, different community composition, and more great scallops *Pecten maximus* compared to sites where towed-fishing occurred. More species were recorded at long-term untowed sites (untowed for 24 years; 16–21 species/tow) and short-term untowed sites (untowed for less than two years; 23–25 species/tow) than at towed sites (8–10 species/tow). Biomass was higher at long-term untowed sites (9.2–9.7 kg/tow) than short-term untowed sites (4.0–8.1 kg/tow); and both were higher than towed sites (0.8–1.5 kg/tow). Community composition at long- and short-term untowed sites (combined) were only 11% similar to that of towed sites. In addition, abundance of great scallops was higher at long-term untowed sites (4–53/tow) and short-term untowed sites (3–15/tow) than at towed sites (0–2/tow). In 1978 a zoned fishery management system was established in a 500 km^2 area, which included a static-gear-only area. In 2002, seven sites were surveyed: two long-term untowed (static-only), two short-term untowed, and three towed sites. Dredges were towed for 10 mins three times at each site (two standard dredges to collect great scallops >100 mm in length; one scientific dredge for other invertebrates). Species were identified and wet-weighed (individuals combined per species).

A site comparison study from 1989–2003 in two sites of soft seabed off the southwest coast of the Isle of Man, Irish Sea (5) found that an area closed to towed fishing gear for 14 years had more and larger great scallops *Pecten maximus* compared to an adjacent fished area. Fourteen years after closure, abundance of scallops was higher in the closed area (14/100 m^2) compared to the fished area (3/100 m^2). In addition, the proportions of older and larger scallops were higher in the closed area (41% over 5-year old; 52% over 130 mm in length) compared to the fished area (5% over 5-year old; 12% over 130 mm). A 2 km^2 exclusion zone was closed to towed fishing gear in 1989 following a bylaw (static gears allowed). Abundance, size, and age of scallops inside and outside the exclusion zone were obtained from a combination of dive surveys and annual dredge surveys carried out during multiple studies between 1989 and 2003 (see paper for details). In the fished area, all surveys were carried out during the closed scallop season June–October). Only data for 2002–2003 were statistically tested.

A replicated, controlled, before-and-after study in 2000–2002 of 14 sites within seagrass beds in the Ria Formosa lagoon, southern Portugal (6) found that ceasing towed gear fishing led to increases in the cover of mobile invertebrates, but not non-moving (sessile) invertebrates, after 10 months. Cover of mobile invertebrates increased after towed-gear fishing stopped (3.9%) compared to before it stopped (1.1%), but not cover of sessile invertebrates (before: 2.8%; after: 2.7%). No changes were reported at sites where experimental fishing continued and at sites never fished (data not provided; no statistical comparisons were made with sites where fishing stopped). The use of towed demersal gears for commercial and recreational purposes is prohibited in the Ria Formosa. Experimental towed gear fishing started in October 2000 at 12 sites (monthly 10 m tow of a beach seine, 9 mm mesh) and stopped at nine of them in September 2001. Cover of sessile and mobile invertebrates (>2.5 cm) was surveyed at all sites and two nearby sites that were never fished during underwater visual surveys (180 m^2/site) before fishing stopped (August–September 2001) and 10 months after (June–July 2002).

A replicated, site comparison study in 2000–2003 in sites of soft seabed on Georges Bank, North Atlantic Ocean, east of Massachusetts, USA (7) found that there were more starfish *Asterias* spp. in areas closed to towed fishing for five to nine years compared to adjacent fished areas, but there was no difference in starfish arm length. Across all years, starfish abundance was higher in the closed areas (0.1–0.6 starfish/m^2), compared to the fished areas (0.0–0.3 starfish/m^2). However, the average arm length of starfish was similar in the closed (20–73 mm) and the fished areas (20–42) and varied between years. In 1994, three areas (17,000 km^2 in total) of Georges Bank (located 13–150 m depth) were closed to towed fishing gear. Portions of the closed areas were re-opened from 1999–2001 for a short-term limited fishery. Between 1999 and 2003, video surveys were undertaken within each closed area and in three areas of Georges Bank opened to fishing. A total of 3,209 stations were video-surveyed, and four 2.8 m^2 video-quadrats/station assessed. All *Asteria* spp. starfish (>2 cm diameter) were counted and their arm lengths measured.

A site comparison study in 2004–2008 in two areas of gravelly and sandy seabed on Georges Bank, northwest Atlantic Ocean, USA (8) found that, 10–14 years after closure, an area closed to commercial towed fishing gear had a higher biomass of invertebrates attached to the seabed (epifauna), but not a higher total abundance or species richness, compared to a fished area. Epifauna biomass was significantly higher in the closed area (33–109 g/L) compared to the fished area (26–57 g/L). Total epifauna abundance was similar in closed (6–15 individuals/L) and fished areas (6–10 individuals/L). The effect of closing commercial fishing on species richness varied with years, but overall across year species richness was similar in both areas (closed: 26–39 species; fished: 32–41 species). An area on Georges Bank was closed to all commercial fishing gear capable of retaining ground fish (trawls, scallop dredges, gill nets and hook gear) in December 1994. Annually between 2004 and 2008, one site in the closed are and one site in an adjacent fished area were surveyed at 45–55 m depth. Epifauna were collected using a dredge (2–3 samples/site/year; 6.4 mm mesh liner), identified, counted, and wet-weighed.

(1) Hoffmann E. & Dolmer P. (2000) Effect of closed areas on distribution of fish and epibenthos. *ICES Journal of Marine Science*, 57, 1310–1314.
(2) Kaiser M.J. Spence F.E. & Hart P.J.B. (2000) Fishing-gear restrictions and conservation of benthic habitat complexity. *Conservation Biology*, 14, 1512–1525.
(3) Hermsen J.M., Collie J.S. & Valentine P.C. (2003) Mobile fishing gear reduces benthic megafaunal production on Georges Bank. *Marine Ecology Progress Series,* 260, 97–108.
(4) Blyth R.E., Kaiser M.J., Edwards-Jones G. & Hart P.J.B. (2004) Implications of a zoned fishery management system for marine benthic communities. *Journal of Applied Ecology*, 41, 951–961.
(5) Beukers-Stewart B.D., Vause B.J., Mosley M.W.J., Rossetti H.L. & Brand A.R. (2005). Benefits of closed area protection for a population of scallops. *Marine Ecology Progress Series,* 298, 189–204.
(6) Curtis J.M., Ribeiro J., Erzini K. & Vincent A.C. (2007) A conservation trade-off? Interspecific differences in seahorse responses to experimental changes in fishing effort. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 17, 468–484.
(7) Marino II M.C., Juanes F. & Stokesbury K.D.E. (2007).Effect of closed areas on populations of sea star *Asterias* spp. on Georges Bank. *Marine Ecology Progress Series,* 347, 39–49.
(8) Smith B.E. Collie J.S. & Lengyel N.L. (2013) Effects of chronic bottom fishing on the benthic epifauna and diets of demersal fishes on northern Georges Bank. *Marine Ecology Progress Series*, 472, 199–217.

Static fishing gear

6.8. Cease or prohibit static fishing gear

- We found no studies that evaluated the effects of ceasing or prohibiting static fishing gear on subtidal benthic invertebrate 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

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear coming into contact with the seabed (Collie *et al*. 2000). Static fishing gear such as pots and traps, although usually considered less damaging than mobile gears, can be locally damaging to the seabed and subtidal benthic invertebrates directly located under or in their vicinity. Ceasing or prohibiting static gears in an area, for instance through bylaws or voluntary agreements (Blyth *et al*. 2002), can remove their direct pressure to subtidal benthic invertebrates and allow them to recolonise and recover naturally over time (Blyth *et al.* 2004). When the cessation of static fishing gear occurs in the context of a marine protected area, the evidence has been summarised under "Habitat protection – Designate a Marine Protected Area and prohibit static fishing gear".

Blyth R.E., Kaiser M.J., Edwards-Jones G. & Hart P.J.B. (2004) Implications of a zoned fishery management system for marine benthic communities. *Journal of Applied Ecology*, 41, 951–961

Blyth R.E., Kaiser M.J., Edwards-Jones G. & Hart P.J.B. (2002) Voluntary management in an inshore fishery has conservation benefits. *Environmental Conservation*, 29, 493–508.

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

Effort and Capacity Reduction

6.9. Establish territorial user rights for fisheries

- **One study** examined the effects of establishing territorial user rights for fisheries on subtidal benthic invertebrate populations. The study was in the South Pacific Ocean[1] (Chile).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Mollusc reproductive success (1 study):** One site comparison study in South Pacific Ocean[1] found that an area with territorial user rights for fisheries had larger-sized and more numerous egg capsules, and more larvae of the Chilean abalone up to 21 months after establishing fishing restrictions compared to an open-access area.

Background

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear coming into contact with the seabed (Collie *et al*. 2000). Territorial user rights are area-based fishing rights which allocate secure, exclusive access to fish in a specified area to groups or individuals (Raemaekers *et al*. 2011). These territorial user rights often have controls on fishing mortality, and fishers are held accountable to comply with these controls. By regulating and limiting fishing effort, establishing territorial user rights can reduce the impact on the seabed, the amount of bycatch, and overall threat to subtidal benthic invertebrates (Gelcich & Donlan 2015; Manríquez & Castilla 2001). Evidence for related interventions is summarised under "Habitat protection – Establish community-based fisheries management".

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

Gelcich S. & Donlan C.J. (2015) Incentivizing biodiversity conservation in artisanal fishing communities through territorial user rights and business model innovation. *Conservation Biology*, 29, 1076–1085.

Manríquez P.H. & Castilla J.C. (2001) Significance of marine protected areas in central Chile as seeding grounds for the gastropod *Concholepas concholepas*. *Marine Ecology Progress Series*, 215, 201–211.
Raemaekers S., Hauck M., Bürgener M., Mackenzie A., Maharaj G., Plagányi É.E. & Britz P.J. (2011) Review of the causes of the rise of the illegal South African abalone fishery and consequent closure of the rights-based fishery. *Ocean & Coastal Management*, 54, 433–445.

A site comparison study in 1993–1994 in two rocky seabed areas in the South Pacific Ocean, central Chile (1) found that an area with territorial user rights for fisheries had larger-sized and more numerous egg capsules, and more larvae of the Chilean abalone *Concholepas concholepas* compared to an open-access area, up to 21 months after establishing fishing restrictions. Egg capsules were bigger in the semi-restricted area (1.9–2.0 cm) than in the open-access area (1.5–1.6 cm). On average, more egg capsules and larvae were produced annually in the semi-restricted area (1993: 69,300 egg capsules/transect, 429 million larvae/transect; 1994: 76,000 egg capsules, 534 million larvae) than in the open-access area (1993: 6,600 egg capsules, 23 million larvae; 1994: 9,900 egg capsules, 34 million larvae). Between January 1993 and December 1994, one diver surveyed a total of 34 transects (90 m^2) across two areas. One area (12 transects in both 1993 and 1994) was under the control of a fishermen's Union group established in 1993 and semi-restricted to fishing (territorial user rights). The other area was an adjacent open-access fishery ground where harvest of the Chilean abalone occurred year-round (six transects in 1993, four transects in 1994). Along each transects, the diver counted and measured Chilean abalone egg capsules, and estimated the number of larvae.

(1) Manríquez P.H. & Castilla J.C. (2001) Significance of marine protected areas in central Chile as seeding grounds for the gastropod *Concholepas concholepas*. *Marine Ecology Progress Series*, 215, 201–211.

6.10. Set commercial catch quotas

- We found no studies that evaluated the effects of setting commercial catch quotas on subtidal benthic invertebrate 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

Many populations of marine subtidal benthic invertebrate species have declined or been depleted due to the multiple threats they are exposed to, including overharvesting (Hobday *et al*. 2000) and unintentional physical damage or catching during other fishing operations (Collie *et al*. 2000). Commercial fishing and harvest quotas (such as Total Allowable Catch) are a means by which many governments and local regulatory bodies regulate biological resources (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. Evidence for the use of catch quotas in recreational fishing is summarised under "Species management – Set recreational catch quotas". Evidence for the use of catch quotas in conjunction with habitat credits system is summarised under "Threat: Biological resource use – Set catch quotas and habitat credits systems".

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.
Hobday A.J., Tegner M.J. & Haaker P.L. (2000) Over-exploitation of a broadcast spawning marine invertebrate: decline of the white abalone. *Reviews in Fish Biology and Fisheries*, 10, 493–514.

6.11. Set habitat credits systems

- We found no studies that evaluated the effects of setting habitat credits systems on subtidal benthic invertebrate 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

Many populations of marine subtidal benthic invertebrate species have declined or been depleted due to the multiple threats they are exposed to, including overharvesting (Hobday *et al*. 2000) and unintentional physical damage or catching during other fishing operations (Collie *et al*. 2000). Habitat credits systems are fisheries management tools aimed to balance economic and environmental values associated with fisheries. In the case of fisheries, the aim is to address specific conservation goals while having minimal effects for the fisheries. A set number of habitat credits (or "individual habitat quotas") are allocated to fishers. Habitat impacts credits are then assigned to specific fishing areas based on their sensitivity to fishing practices; the more sensitive the area, the more habitat credits it will require from the fisher to go and fish there. Setting habitat credits systems for specific exploited areas, can potentially incentivise responsible fishing practices by constraining fishers to a set number of credits or shares of the habitat, while allowing them to change their behaviour (where, when, and how much they fish) (Bastleer *et al.* 2017). This may reduce the pressure on particularly sensitive areas and their associated species. Direct evidence is limited, but indirect evidence using modelling approaches have shown that habitat credit systems could reduce benthic impacts (Bastleer *et al.* 2017).

Evidence for the use of habitat credits system in conjunction with catch quotas is summarised under "Threat: Biological resource use – Set catch quotas and habitat credits systems".

Batsleer J., Marchal P., Vaz S., Vermard V, Rijnsdorp A.D., Poos J.J. (2018) Exploring habitat credits to manage the benthic impact in a mixed fishery. *Marine Ecology Progress Series*, 586, 167–179.
Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.
Hobday A.J., Tegner M.J. & Haaker P.L. (2000) Over-exploitation of a broadcast spawning marine invertebrate: decline of the white abalone. *Reviews in Fish Biology and Fisheries*, 10, 493–514.

6.12. Set commercial catch quotas and habitat credits systems

- We found no studies that evaluated the effects of setting commercial catch quotas and habitat credits systems on subtidal benthic invertebrate 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

Many populations of marine subtidal benthic invertebrate species have declined or been depleted due to the multiple threats they are exposed to, including overharvesting (Hobday *et al*. 2000) and unintentional physical damage or catching during other fishing operations (Collie *et al*. 2000). Commercial fishing and harvest quotas (such as Total Allowable Catch) are a means by which many governments and local regulatory bodies regulate biological resources (species stocks). Habitat credits systems are fisheries management tools aimed to balance economic and environmental values associated with fisheries. In the case of fisheries, the aim is to address specific conservation goals while having minimal effects for the fisheries. A set number of habitat credits (or "individual habitat quotas") are allocated to fishers. Habitat impacts credits are then assigned to specific fishing areas based on their sensitivity to fishing practices; the more sensitive the area, the more habitat credits it will require from the fisher to go and fish there.

Setting habitat credits systems for specific exploited areas, can potentially incentivise responsible fishing practices by constraining fishers to a set number of credits or shares of the habitat, while allowing them to change their behaviour (where, when, and how much they fish) (Bastleer *et al.* 2017). Setting catch quotas in conjunction with habitat credits systems for specific fisheries and areas (for instance cod in the English Channel, Bastleer *et al*. 2017), can potentially reduce the pressure on particularly sensitive areas and their associated species. Direct evidence is limited, but indirect evidence using modelling approaches have shown that using catch quotas in conjunction with habitat credit systems could reduce benthic impacts (Bastleer *et al.* 2017).

Evidence for the use of habitat credits system not in conjunction with catch quotas is summarised under "Threat: Biological resource use – Set habitat credits systems", while evidence for the use of catch quotas alone is summarised under "Threat: Biological resource use – Set catch quotas".

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

Hobday A.J., Tegner M.J. & Haaker P.L. (2000) Over-exploitation of a broadcast spawning marine invertebrate: decline of the white abalone. *Reviews in Fish Biology and Fisheries*, 10, 493–514.

6.13. Limit the number of fishing days

- We found no studies that evaluated the effects of limiting the number of fishing days on subtidal benthic invertebrate 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

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear coming into contact with the seabed (Collie *et al*. 2000). The number of fishing days could be limited to reduce fishing effort in an area, thereby reducing the threat to subtidal benthic invertebrates, and may allow time for them to potentially recover during the non-fishing days.

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

6.14. Limit the number of fishing vessels

- We found no studies that evaluated the effects of limiting the number of fishing vessels on subtidal benthic invertebrate 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

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear coming into contact with the seabed (Collie *et al*. 2000). The number of fishing vessels could be limited to reduce fishing effort in an area (Huan & Chuang 2010), thereby potentially reducing the impact on the seabed, the amount of unwanted catch, and overall threat to subtidal benthic invertebrates.

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

Huang H. W. & Chuang C. T. (2010). Fishing capacity management in Taiwan: Experiences and prospects. *Marine Policy*, 34, 70–76.

6.15. Limit the number of traps per fishing vessels

- We found no studies that evaluated the effects of limiting the number of traps per fishing vessels on subtidal benthic invertebrate 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

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear coming into contact with the seabed (Collie *et al*. 2000). Traps or pots are often used to fish for crabs or lobsters and consist of structures into which species of commercial interest enter through funnels, which encourage entry but limit escape. Trap fishery can negatively impact benthic invertebrates by accidentally catching them while in use (Öndes *et al.* 2017), or when they are lost or abandoned ("ghost fishing" Maselko *et al.* 2013). The number of traps per fishing vessels could be limited to reduce fishing effort in an area (Acheson 1998), thereby potentially reducing the amount of unwanted catch, and overall threat to subtidal benthic invertebrates. Evidence related to interventions aimed at mitigating ghost fishing is summarised under "Threat: Pollution – Use biodegradable panels in fishing pots" and "Recover lost fishing gear".

Acheson J. (1998) Lobster trap limits: A solution to a communal action problem. *Human Organization*, 57, 43–52.

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

Maselko J., Bishop G. & Murphy P. (2013) Ghost fishing in the Southeast Alaska commercial Dungeness crab fishery. *North American Journal of Fisheries Management*, 33, 422–431.
Öndes F., Kaiser M.J. & Murray L.G. (2017) Fish and invertebrate by-catch in the crab pot fishery in the Isle of Man, Irish Sea. *Journal of the Marine Biological Association of the United Kingdom*, 98, 1–13.

6.16. Limit the density of traps

- We found no studies that evaluated the effects of limiting the density of traps on subtidal benthic invertebrate 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

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear coming into contact with the seabed (Collie *et al*. 2000). Traps or pots are often used to fish for crabs or lobsters and consist of structures into which species of commercial interest enter through funnels, which encourage entry but limit escape. Trap fishery can negatively impact benthic invertebrates by accidentally catching them while in use (Öndes *et al.* 2017), or when they are lost or abandoned ("ghost fishing" Maselko *et al.* 2013). The number of traps in a given area (density of traps) could be limited to reduce local fishing effort (Acheson 1998; Miller 1976), thereby reducing the amount of unwanted catch, and overall threat to subtidal benthic invertebrates. Related evidence is summarised under "Threat: Biological resource use – Limit the number of traps per fishing vessels". Evidence related to interventions aimed at mitigating ghost fishing is summarised under "Threat: Pollution – Use biodegradable panels in fishing pots" and "Recover lost fishing gear".

Acheson J. (1998) Lobster trap limits: A solution to a communal action problem. *Human Organization*, 57, 43–52.
Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.
Maselko J., Bishop G. & Murphy P. (2013) Ghost fishing in the Southeast Alaska commercial Dungeness crab fishery. *North American Journal of Fisheries Management*, 33, 422–431.
Miller, R.J. (1976) North American crab fisheries: regulations and their rationales. *Fishery Bulletin*, 74, 623–633.
Öndes F., Kaiser M.J. & Murray L.G. (2017) Fish and invertebrate by-catch in the crab pot fishery in the Isle of Man, Irish Sea. *Journal of the Marine Biological Association of the United Kingdom*, 98, 1–13.

6.17. Install physical barriers to prevent trawling

- **One study** examined the effects of installing physical barriers to prevent trawling on subtidal benthic invertebrate populations. The study was in the Bay of Biscay[1] (Spain).

COMMUNITY RESPONSE (1 STUDY)

- **Overall community composition (1 study):** One before-and-after study in the Bay of Biscay[1] found that one to four years after installing artificial reefs as physical barriers to prevent trawling invertebrate community composition changed.

POPULATION RESPONSE (1 STUDY)

- **Overall abundance (1 study):** One before-and-after study in the Bay of Biscay[1] found that one to four years after installing artificial reefs as physical barriers to prevent trawling overall invertebrate biomass increased.
- **Echinoderm abundance (1 study):** One before-and-after study in the Bay of Biscay[1] found that one to four years after installing artificial reefs as physical barriers to prevent trawling the biomass of sea urchins and starfish increased.
- **Molluscs abundance (1 study):** One before-and-after study in the Bay of Biscay[1] found that one to four years after installing artificial reefs as physical barriers to prevent trawling the biomass of gastropods (sea snails), of one species of cuttlefish, and of two species of octopus increased.

Background

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear coming into contact with the seabed (Collie *et al*. 2000). Some habitats, such as coral reefs and seagrass meadows, are particularly vulnerable to trawling gears. Trawling can be discouraged in certain areas through the positioning of physical barriers, such as concrete blocks or other artificial reefs, to make trawling unfeasible without damaging trawl nets (Liu *et al*. 2011). This may limit local trawling (and other mobile fishing) effort in the area, thereby reducing the impact on the seabed, the amount of bycatch, and overall threat to subtidal benthic invertebrates. When this intervention occurs within a marine protected area, evidence has been summarised under "Designate a Marine Protected Area and install physical barriers to prevent illegal trawling". Evidence for related interventions is summarised under "Habitat restoration and creation – Place anthropogenic installations (e.g: windfarms) in an area such that they create artificial habitat and reduce the level of fishing activity".

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

Liu X.-S., Xu W.-Z., Cheung S.G. & Shin P.K.S. (2011) Response of meiofaunal community with special reference to nematodes upon deployment of artificial reefs and cessation of bottom trawling in subtropical waters, Hong Kong. *Marine Pollution Bulletin,* 63, 376–384.

A before-and-after study in 1998–2007 in one area of soft seabed in the Cantabrian Sea, southern Bay of Biscay, Spain (1) found that one to four years after installing barriers to prevent illegal trawling the biomass of invertebrates increased, and species community composition changed. Total invertebrate biomass was higher after one (3 kg/ha) and four years (7 kg/ha), compared to before installation (0–1 kg/ha). There were increases in the biomass of sea urchins (before: 12; after: 3,150 g/ha), common octopus *Octopus vulgaris* (before: 222; after: 920 g/ha), starfish (before: 9; after: 78 g/ha), gastropods (before: 18; after: 50 g/ha), cuttlefish *Sepia* spp. (before: 56; after: 131 g/ha), and curled octopus *Eledone cirrhosa* (before: 9; after: 18 g/ha). Invertebrate community composition was different before and after deployment (results presented as graphical analyses). Bottom trawling in the area was prohibited at depths <100 m by local legislation, but illegal trawling was common. To prevent it, artificial reefs (groups of concrete blocks 2 km apart; numbers not specified) were deployed in 2003 at 80–85 m depth. One sampling station near each group of blocks (sandy seabed without blocks) was surveyed annually in October in 1998–2007 (survey methods not specified).

(1) Serrano A. Rodríguez-Cabello C. Sánchez F. Velasco F. Olaso I. & Punzón A. (2011) Effects of anti-trawling artificial reefs on ecological indicators of inner shelf fish and invertebrate communities in the

Cantabrian Sea (southern Bay of Biscay). *Journal of the Marine Biological Association of the United Kingdom,* 91, 623–633.

6.18. Introduce catch shares

• We found no studies that evaluated the effects of introducing catch shares on subtidal benthic invertebrate 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 shares are a type of fisheries management system whereby a secure share of targeted species or fishing area is given to individual fishermen, communities or fisheries associations (Costello *et al.* 2008). This normally involves the division of allocated catch quotas amongst fishery participants, who can then catch a certain amount of targeted species each year and are responsible for not exceeding that amount. Shares can sometimes be bought or sold. With a secure share of the catch, there can be reduced competition between fishery participants and focus may shift from maximising volume to maximising value, which may reduce fishing effort in the area, thereby potentially reducing the impact on the seabed, the amount of unwanted catch, and overall threat to subtidal benthic invertebrates.

Costello C., Gaines S.D. & Lynham J. (2008) Can catch shares prevent fisheries collapse? *Science*, 321, 1678–1681.

6.19. Purchase fishing permits and/or vessels from fishers

• We found no studies that evaluated the effects of purchasing fishing permits and/or vessels from fishers on subtidal benthic invertebrate 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

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear coming into contact with the seabed (Collie *et al*. 2000). Schemes can be set up where fishing permits and/or vessels are purchased from fishers. This can reduce fishing effort in an area due to fewer fishing boats operating (Gleason *et al.* 2013), thereby potentially reducing the impact on the seabed, the amount of unwanted catch, and overall threat to subtidal benthic invertebrates.

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.
Gleason M., Feller E.M., Merrifield M., Copps S., Fujita R.O.D., Bell M., Rienecke S. & Cook C. (2013) A transactional and collaborative approach to reducing effects of bottom trawling. *Conservation Biology,* 27, 470–479.

6.20. Eliminate fisheries subsidies that encourage overfishing

- We found no studies that evaluated the effects of eliminating fisheries subsidies that encourage overfishing on subtidal benthic invertebrate 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

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear coming into contact with the seabed (Collie *et al*. 2000). Overfishing is thought to be encouraged by fisheries subsidies, which provides funds that go towards building boats, and paying for fuel and fishing gear. By eliminating fisheries subsidies that encourage overfishing (Sumaila & Pauly 2007) fishing effort can be reduced in an area, thereby potentially reducing the impact on the seabed, the amount of unwanted catch, and overall threat to subtidal benthic invertebrates.

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.
Sumaila U.R. & Pauly D. (2007) All fishing nations must unite to cut subsidies. *Nature*, 450, 945.

Reduce Unwanted catch, Discards and Impacts on seabed communities

6.21. Set unwanted catch quotas

- We found no studies that evaluated the effects of setting unwanted catch quotas on subtidal benthic invertebrate 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 fisheries terms, unwanted catch (sometimes referred to as "bycatch", although the exact meaning of this term varies) are species whose captures were unintentional or unwanted. For instance, these can include undersize or over quota for species of commercial value, but also species that hold no commercial value. Here, we only consider unintentionally captured species of no commercial value. Some fishing practices can lead to considerable amounts of benthic invertebrate unwanted catch (Davies *et al*. 2009). Unwanted catch 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 potentially reduce fishing in an area, thereby reducing the impact on the seabed, the amount of unwanted catch, and overall threat to subtidal benthic invertebrates.

Davies R.W.D., Cripps S.J., Nickson A. & Porter G. (2009) Defining and estimating global marine fisheries unwanted catch. *Marine Policy*, 33, 661–672.

6.22. Use hook and line fishing instead of other fishing methods

- We found no studies that evaluated the effects of using hook and lime fishing instead of other fishing methods on subtidal benthic invertebrate 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

Hook and line fishing is a term used for a range of fishing methods that use short fishing lines with hooks. Hook and line fishing is more selective than other types of fishing and has little impact on the seabed. In addition, unwanted catch species can often be returned to the sea undamaged because the lines are only in place for a short time. These methods also reduce direct contact with the seabed, any unintentional physical harm and disturbances, and potentially reduce the amount of unwanted catch. For evidence of the effect of this intervention within a marine protected area, see "Habitat protection – Designate a Marine Protected Area and only allow the use hook and line fishing instead of other fishing methods".

6.23. Use a midwater/semi-pelagic trawl instead of bottom/demersal trawl

- **One study** examined the effects of using a semi-pelagic trawl instead of a demersal trawl on subtidal benthic invertebrates. The study was in the Indian Ocean[1] (Australia).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Overall abundance (1 study):** One replicated, controlled, study in the Indian Ocean[1] found that fishing with a semi-pelagic trawl did not reduce the abundance of large sessile invertebrates, which was similar to non-trawled plots, but a demersal trawl did.

OTHER (1 STUDY)

- **Commercial catch abundance (1 study):** One replicated, controlled, study in the Indian Ocean[1] found that fishing with a semi-pelagic trawl reduced the abundance of retained commercially targeted fish compared to fishing with a demersal trawl.

Background

Trawling is a method of fishing that involves pulling a fishing net (trawl) through the water behind one or more vessels. Semi-pelagic trawls (also referred to as midwater trawls) are types of trawls where the nets are towed through the water above the seabed, whereas demersal trawls (also referred to as bottom trawls) tow their nets along, or close to, the seabed. Semi-pelagic trawl gear does not come into contact with the seabed, resulting in less damage to the seabed. Using a semi-pelagic trawl instead of a demersal trawl may potentially reduce the impact on the seabed and disturbances or damages to subtidal benthic invertebrates (Moran & Stephenson 2000).

Moran M.J. & Stephenson P.C. (2000) Effects of otter trawling on macrobenthos and management of demersal scalefish fisheries on the continental shelf of north-western Australia. *ICES Journal of Marine Science*, 57, 510–516.

A replicated, controlled study (date of study unspecified) in six seabed plots in the Indian Ocean, north-western Australia (1) found that fishing with a semi-pelagic trawl did not reduce invertebrate abundance to the extent that fishing with a standard demersal trawl did. Following fishing with a semi-pelagic trawl, abundance of large sessile invertebrates (>20 cm) did not change and was similar to non-trawled plots, whereas following each fishing event with a demersal trawl their abundance was reduced by approximately 16% (data presented on a logarithm scale). The semi-pelagic trawl caught fewer commercially targeted fish than the demersal trawl. Two types of otter trawls were tested; a semi-pelagic trawl, deployed approximately 15 cm above the seabed (no contact), and a standard demersal trawl towed along the seabed (in contact). An area of seabed at 50–55 m depth that had never been trawled was divided into six 360 x 925 m plots: two plots/gear type and two non-trawled plots. Each trawled plot was trawled four times. Invertebrates attached to the seabed (>20 cm, reported as being mostly sponges, soft corals, and gorgonians) were counted from video camera recordings before and after each fishing event in trawled and non-trawled plots.

(1) Moran M.J. & Stephenson P.C. (2000) Effects of otter trawling on macrobenthos and management of demersal scalefish fisheries on the continental shelf of north-western Australia. *ICES Journal of Marine Science*, 57, 510–516.

6.24. Modify the design of dredges

- **Six studies** examined the effects of modifying the design of dredges on subtidal benthic invertebrate populations. Four were in the North Atlantic Ocean[1,3a-c,4] (Portugal) and two were in the Irish Sea[2,5] (Isle of Man).

COMMUNITY RESPONSE (1 STUDY)

- **Unwanted catch overall composition (1 study):** One replicated, controlled, study in the Irish Sea[5] found that a new design of scallop dredge caught a similar species composition of unwanted catch to a traditional dredge.

POPULATION RESPONSE (3 STUDIES)

- **Overall abundance (2 studies):** One of two controlled studies in the North Atlantic Ocean[4] and in the Irish Sea[5] found that a new dredge design damaged or killed fewer invertebrates left in the sediment tracks following dredging[4]. The other[5] found no difference in total invertebrate abundance or biomass living in or on the sediment tracks following fishing with two dredge designs.
- **Unwanted catch overall abundance (2 studies):** Two controlled studies (one replicated) in the North Atlantic Ocean[1] and the Irish Sea[5] found that a modified or a new design of bivalve dredge caught less unwanted catch compared to traditional unmodified dredges.
  Unwanted catch condition (6 studies): Six controlled studies (one replicated and paired, four replicated) in the North Atlantic Ocean[1,3a-c,4] and the Irish Sea[2] found that new or modified bivalve dredges damaged or killed similar proportions of unwanted catch (retained and/or escaped) compared to traditional or unmodified designs, three of which also found that they did not reduce the proportion of damaged or dead unwanted crabs (retained and/or escaped)[3a-c].

OTHER (1 study)

- **Commercial catch abundance (1 study):** One replicated, controlled, study in the Irish Sea[5] found that a new dredge design caught a similar amount of commercially targeted queen scallops compared to a traditional dredge.

Background

Dredging, for instance for bivalves, normally involves towing a heavy steel frame along the seabed, which negatively impacts subtidal benthic invertebrates directly, due to physical damage and the retention of unwanted invertebrate catch, and indirectly by changes to the seabed structure and topography, such as creating dredge tracks (Currie & Parry 1996). Dredge design can be modified in order to reduce or remove negative impacts on subtidal benthic invertebrates, such as changing the tooth spacing or mesh size, or reducing sediment penetration or bottom-contact (Frandsen *et al.* 2015). Evidence for other interventions related to the use of dredges are summarised under "Threat: Biological resource use – Cease or prohibit dredging", "Use lower water pressure during hydraulic dredging", "Hand harvest instead of using a dredge", " Use an otter trawl instead of a dredge", "Use alternative means of getting mussel seeds rather than dredging from natural mussel beds", "Use an otter trawl instead of a dredge", and in "Species management – Cease or prohibit the harvest of scallops".

Currie D.R. & Parry G.D. (1996) Effects of scallop dredging on a soft sediment community: a large-scale experimental study. *Marine Ecology Progress Series*, 134, 131–150.

Frandsen R.P., Eigaard O.R., Poulsen L.K., Tørring D., Stage B., Lisbjerg D. & Dolmer P. (2015) Reducing the impact of blue mussel (*Mytilus edulis*) dredging on the ecosystem in shallow water soft bottom areas. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 25, 162–173

A controlled study in 1999 in one sandy area in the North Atlantic Ocean, off the southwest coast of Portugal (1) found that a modified bivalve dredge caught less unwanted invertebrate catch, but damaged or killed similar proportions of unwanted invertebrates, compared to a traditional unmodified dredge. The proportion of unwanted individuals/tow was lower for the modified dredge (30–35%), compared to the traditional dredge (42–62%). This pattern was also true when looking at the proportion by weight (modified: 24–26%; traditional: 25–47%). However, the modified dredge did not have statistically lower proportion of damaged invertebrates overall (6–9%) compared to traditional dredges (7–14%), or lower proportion of dead invertebrates (modified: 6–8%; traditional: 6.5–11%), but this effect varied with species (see paper for details). The modified dredge had a metallic grid for retaining bivalves, compared to the traditional dredge which had a net bag. All species other than the commercially targeted smooth clam *Callista chione* were considered to be unwanted catch. In March, 12 tows/design were investigated at 8–10 m depth. A net bag was fitted to the end of each dredge to retain the caught organisms that would otherwise escape through the dredge mesh. For each dredge design, invertebrates were identified, counted, and given a score of 1–4 according to the amount of damage (1 = perfect condition, 4 = dead).

A replicated, paired, controlled study in 1994–1995 in 13 soft seabed sites in the northern Irish Sea, Isle of Man (2) found that scallop dredges with shorter teeth caused similar damage to unwanted invertebrate catch, compared to dredges with longer teeth. The damage sustained by unwanted invertebrates was similar when caught in the dredge with shorter and longer teeth (results not shown). A modified dredge design, with shorter teeth and smaller belly ring, was compared to a traditional design (Newhaven with spring-teeth). In 1994 and 1995, up to 13 fishing grounds were surveyed in June and October (at the start and end of the closed fishing season for great scallops *Pecten maximus*). In each area, one boat simultaneously towed a group of four modified dredges, and a group of four traditional dredges over 2 nm (one group on either side). Unwanted invertebrate catch (crabs, starfish, urchins, whelks, bivalves, hermit crabs, octopus) was sorted to species level, counted, and given a damage score (1= no visible damage, 4= crushed/dead).

A replicated, controlled study in 1999 in one area of sandy seabed in the North Atlantic Ocean, off northwestern Portugal (3a) found that modifying dredge tooth spacing did not reduce the proportion of damaged or dead individuals (unwanted catch and escapees) for either the overall combined invertebrate and fish community or for crabs. The proportions of individuals in the overall community that entered the dredge and were damaged or dead were similar using a 2 cm (damaged: 3–5%, dead: 1–1.5%), 4 cm (damaged: 3–4%, dead: 0–1%) or 6 cm (damaged: 4–8%, dead: 1%) tooth spacing design. The proportions of crabs that were damaged or dead were similar using a 2 cm (damaged: 8–25%, dead: 4–11%), 4 cm (damaged: 7–19%, dead: 1–9%) or 6 cm (damaged: 7–29%, dead:1–20%) tooth spacing design. Three tooth-spacing designs (2, 4 and 6 cm) were compared. In July, two bivalve dredges with different designs were towed simultaneously side-by-side at 8–10 m depth (three tows/design; 15 min/tow). A net bag was fitted to the end of each dredge to retain the caught organisms that would otherwise escape through the dredge mesh. For each dredge design, catches were sorted by species group, counted and given a score of 1–5 according to the amount of damage (1 = good condition, 5 = crushed/dead). The effect of tooth spacing was examined for the overall unwanted community (invertebrates and fish) and for crabs.

A replicated, controlled study in 1999 in one area of sandy seabed in the North Atlantic Ocean, off northwestern Portugal (3b) found that modifying the net mesh size on a dredge did not reduce the proportion of damaged or dead individuals (unwanted catch and escapees) for either the overall combined invertebrate and fish community or for crabs. The proportions of individuals in the overall community that entered the dredge and were damaged or dead were similar using a 35 mm (damaged: 3–8%, dead: 1%), 40 mm (damaged: 4–5%, dead: 1%) or 50 mm (damaged: 3–5%, dead: 0–1.5%) mesh size design. The proportions of crabs that were damaged or dead were similar using a 35 mm (damaged: 8–24%, dead: 4–20%), 40 mm (damaged: 10–29%, dead: 5–8%), or 50 mm mesh size design (damaged: 7–25%, dead: 1–11%). Three mesh sizes (35 mm, 40 mm and 50 mm) were compared. In July, two bivalve dredges with different designs were towed simultaneously side-by-side at 8–10 m depth (three tows/design; 15 min/tow). A net bag was fitted to the end of each dredge to retain the caught organisms that would otherwise escape through the dredge mesh. For each dredge design, catches were sorted by species group, counted and given a score of 1–5 according to the amount of damage (1 = good condition, 5 = crushed/dead). The effect of mesh size was examined for the overall unwanted community (invertebrates and fish) and for crabs.

A replicated, controlled study in 1999 in one area of sandy seabed in the North Atlantic Ocean, off northwestern Portugal (3c) found that modifying the tooth spacing and net mesh size on dredge did not reduce the proportion of damaged or dead individuals (unwanted catch and escapees) for either the overall combined invertebrate and fish community or for crabs. The proportions of individuals in the overall community that entered the dredge and were damaged or dead were similar for the nine designs tested (damaged: 3–8%, dead: 0–1.5%). The proportions of crabs that were damaged or dead were similar for the nine designs tested (damaged: 7–29%, dead: 1–20%). Nine combinations of three mesh sizes (35, 40 and 50 mm) and three tooth-spacings (2, 4 and

6 cm) were compared. In July, two bivalve dredges with different designs were towed simultaneously side-by-side at 8–10 m depth (three tows/design; 15 min/tow). A net bag was fitted to the end of each dredge to retain the caught organisms that would otherwise escape through the dredge mesh. For each dredge design, catches were sorted by species group, counted and given a score of 1–5 according to the amount of damage (1 = good condition, 5 = crushed/dead). The effect of mesh size was examined for the overall community (invertebrates and fish) and for crabs.

A replicated, controlled study in 2001 in one area of sandy seabed in the North Atlantic Ocean, off southwestern Portugal (4) found that a new dredge design with a shorter mouth did not reduce the proportion of damaged or dead invertebrates caught with the dredge, compared to two traditional dredge designs, but damaged and killed lower proportions of invertebrates left in the tracks following dredging. The proportions of individuals that entered the dredge and were damaged or dead were similar using the new design (damaged: 5%, dead: 5%), a traditional design with a long mouth (damaged: 3%, dead: 3%) and another traditional design with a short mouth (damaged: 7%, dead: 6%). However, the proportion of invertebrates left in the tracks following dredging were lower using the new design (damaged: 17%, dead: 17%), compared to the long-mouthed traditional design (damaged: 42 %, dead: 29%) or the short-mouth traditional design (damaged: 26%, dead: 18%). Three dredge designs were compared: a new design with a shorter mouth and metallic grid instead of a net bag to retain the catch, a traditional design with a long mouth and more teeth, and a traditional design with a short mouth ("north dredge"). A total of 12 tows (4/design; 5 min/tow) were undertaken in June at 8–10 m depth. A net bag was fitted to the end of each dredge to retain the caught organisms that would otherwise escape through the dredge mesh. Divers also sampled the sediment in the dredge tracks after each tow to assess the proportion of invertebrates not caught but left damaged or dead due to dredging (54 quadrats/tow; extracted using a 5 mm mesh sieve). All invertebrates were identified, counted, weighed and given a damage score (1= in good condition, 4= crushed/dead).

A replicated, controlled, study (date unspecified) in an area of sandy seabed in the north Irish Sea, Isle of Man (5) found that a new design of scallop dredge caught similar species of unwanted invertebrates and fish, but in lower amounts, compared to a traditional scallop dredge. Overall unwanted species composition (invertebrates and fish) was similar between the new and the traditional dredge (composition data presented as graphical analyses). Unwanted catch from both dredges was reported to be dominated by invertebrates. The new dredge design caught fewer unwanted invertebrates and fish (23 individuals/1,000 m^2) than the traditional dredge (59). In addition, following fishing impacts, there were no changes in total invertebrate abundance and biomass living in or on the sediments for any of the gears (raw data not presented). The new dredge design caught similar amount of commercially targeted queen scallops *Aequipecten opercularis* (48 scallops/1,000 m^2) compared to the traditional dredge (15). Two queen scallop dredges were compared: a new dredge design with a rubber lip instead of traditional teeth, and a traditional Newhaven dredge. The study site was subdivided into eight trawling lanes (40 m wide, 1 nm long) in 20–23 m water depth. Each fishing lane was allocated to one gear design (4 lanes/design). Commercial and unwanted catches (invertebrates and fish) were sorted, identified, counted and weighed. Before, and seven days after fishing trials, invertebrates (size unspecified) were sampled in each lane using a 2-m beam trawl (5-min tow; 6 tows/lane) and a sediment grab (0.1 m^2; 6 grabs/lane).

(1) Gaspar M.B., Dias M.D., Campos A., Monteiro C.C., Santos M.N., Chicharo A. & Chicharo L. (2001) The influence of dredge design on the catch of *Callista chione* (Linnaeus, 1758). *Hydrobiologia,* 465, 153–167.
(2) Veale L.O. Hill A.S. Hawkins S.J. & Brand A.R. (2001) Distribution and damage to the by-catch assemblages of the northern irish sea scallop dredge fisheries. *Journal of the Marine Biological Association of the United Kingdom,* 81, 85–96.
(3a-c) Gaspar M.B., Leitão F., Santos M.N., Sobral M., Chícharo L., Chícharo A. & Monteiro C.C. (2002) Influence of mesh size and tooth spacing on the proportion of damaged organisms in the catches of the Portuguese clam dredge fishery. *ICES Journal of Marine Science,* 59, 1228–1236.
(4) Gaspar M.B., Leitão F., Santos M.N., Chícharo L., Dias, M.D., Chícharo, A., & Monteiro C.C. (2003) A comparison of direct macrofaunal mortality using three types of clam dredges. *ICES Journal of Marine Science,* 60, 733–742.
(5) Hinz H., Murray, L.G., Malcolm F.R. & Kaiser M.J. (2012) The environmental impacts of three different queen scallop (*Aequipecten opercularis*) fishing gears. *Marine Environmental Research,* 73, 85–95.

6.25. Use lower water pressure during hydraulic dredging

- We found no studies that evaluated the effects of using lower water pressure during hydraulic dredging on subtidal benthic invertebrate 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 species, such as bivalves, can be fished using hydraulic dredges (Moschino *et al.* 2003). Using a hydraulic dredge can negatively affect subtidal benthic invertebrates due to direct physical damage, and changes to the seabed structure and topography (Gilkinson *et al.* 2003). Using a lower water pressure may potentially result in less damage or stress to organisms and theseabed.

Moschino V., Deppieri M. & Marin M.G. (2003) Evaluation of shell damage to the clam *Chamelea gallina* captured by hydraulic dredging in the Northern Adriatic Sea. *ICES Journal of Marine Science*, 60, 393–401.
Gilkinson K.D., Fader G.B.J., Gordon Jr D.C., Charron R., McKeown D., Roddick D., Kenchington E.L.R., MacIsaac K., Bourbonnais C., Vass P. & Liu Q. (2003) Immediate and longer-term impacts of hydraulic clam dredging on an offshore sandy seabed: effects on physical habitat and processes of recovery. *Continental Shelf Research*, 23, 1315–1336.

6.26. Hand harvest instead of using a dredge

- **Two studies** examined the effects of hand harvesting instead of using a dredge on subtidal benthic invertebrate populations. Both were in San Matías Gulf, South Atlantic Ocean[1,2] (Argentina).

COMMUNITY RESPONSE (2 STUDIES)

- **Unwanted catch community composition (1 study):** One replicated, controlled study in San Matías Gulf[2] found that, when harvesting mussels, the community composition of the unwanted catch was similar by hand harvesting and by using a dredge.

- **Unwanted catch richness/diversity (1 study):** One replicated, controlled study in San Matías Gulf[1] found that, when harvesting mussels, hand harvesting caught fewer species of unwanted catch compared to using a dredge.

POPULATION RESPONSE (2 STUDIES)

- **Unwanted catch abundance (1 study):** One replicated, controlled study in San Matías Gulf[1] found that, when harvesting mussels, hand harvesting caught fewer unwanted sea urchins and brittle stars compared to using a dredge.
- **Unwanted catch condition (1 study):** One replicated, controlled study in San Matías Gulf[2] found that, when harvesting mussels, the damage caused to unwanted sea urchins and brittle stars was similar by hand harvesting and by using a dredge.

OTHER 1 STUDY)

- **Commercial catch abundance (1 study):** One replicated, controlled study in San Matías Gulf[2] found that more commercially targeted mussels were caught by hand harvesting than by using a dredge.

Background

Dredging, for instance for bivalves, normally involves towing a heavy steel frame along the seabed, which negatively impacts subtidal benthic invertebrates due to direct physical damage, and changes to the seabed structure and topography (Currie & Parry 1996). Hand harvesting, (for instance using hand-pushed rakes, hand-dredge, dip nets, harvesting-knife, or direct manual harvesting) instead of dredging can have fewer negative impacts on subtidal benthic invertebrates and the surrounding seabed (Bishop *et al.* 2005; Leitão & Gaspar 2007; Narvarte *et al*. 2011).

Bishop M., Peterson C.H., Summerson H.C. & Gaskill D. (2005) Effects of harvesting methods on sustainability of a bay scallop fishery: dredging uproots seagrass and displaces recruits. *Fishery Bulletin*, 103, 712–719.

Currie D.R. & Parry G.D. (1996) Effects of scallop dredging on a soft sediment community: a large-scale experimental study. *Marine Ecology Progress Series*, 134, 131–150.

Leitão F.M.S. & Gaspar M.B. (2007) Immediate effect of intertidal non-mechanised cockle harvesting on macrobenthic communities: a comparative study. *Scientia Marina*, 71, 723–733.

Narvarte M. González R. Medina A. & Avaca M.S. (2011) Artisanal dredges as efficient and rationale harvesting gears in a Patagonian mussel fishery. *Fisheries Research*, 111, 108–115.

A replicated, controlled study in 2007 on a mussel bed in the San Matías Gulf, South Atlantic Ocean, Argentina (1 – same experimental set-up as 2) found that hand-harvesting mussels caught fewer unwanted species including fewer unwanted sea urchins and brittle stars than with standard artisanal dredges. In total, hand-harvesting caught 27 species of unwanted catch, while dredging caught 47. The percentages of unwanted sea urchins *Arbacia dufresnei* and brittle stars *Ophioploccus januarii* caught (% by numbers of total catch) were lower by hand-harvesting (sea urchins: 2%; brittle stars: 32%) than dredging (sea urchins: 9%, brittle stars: 68%). More commercially targeted mussels were caught by hand-harvesting (76%) than the dredge (57% of total catch). Nineteen tows (5 min duration) were conducted in May 2007 on the mussel bed at 14–20 m depth with a standard artisanal dredge (1.6 m mouth width, 80 mm net bag mesh size). Four 40 kg commercial bags of catch hand-harvested by divers in the same area were obtained for comparison. All species were sorted (mussels; unwanted catch), counted, weighed and identified. Average proportions of mussels and unwanted catch (mostly invertebrates) were estimated for each sample. Apart from mussels, sea urchins and brittle stars dominated all samples.

A replicated, controlled study in 2007 on a mussel bed in the San Matías Gulf, South Atlantic Ocean, Argentina (2; same experimental set-up as 1) found that hand-harvesting mussels caught a similar community composition of unwanted catch, and damaged similar numbers of unwanted sea urchins or brittle stars, compared to standard artisanal dredges. The percentages of total sea urchins *Arbacia dufresnei* and brittle stars *Ophioploccus januarii* that were damaged (lightly or severely) were similar by hand-harvesting (sea urchins: 67%; brittle stars: 65%) and dredging (sea urchins: 76%, brittle stars: 75%). Nineteen tows (5 min duration) were conducted in May 2007 on the mussel bed at 14–20 m depth with a standard artisanal dredge (1.6 m mouth width, 80 mm net bag mesh size). Four 40 kg commercial bags of catch hand-harvested by divers in the same area were obtained for comparison. All species were sorted (mussels; unwanted catch), counted, weighed and identified. Average proportions of mussels and unwanted catch (mostly invertebrates) were estimated for each sample. Apart from mussels, sea urchins and brittle stars dominated all samples, and were placed into damage categories: undamaged, lightly damaged or severely damaged (combined under 'damaged').

(1) Narvarte M., González R., Medina A. & Avaca M.S. (2011) Artisanal dredges as efficient and rationale harvesting gears in a Patagonian mussel fishery. *Fisheries Research,* 111, 108–115.
(2) Narvarte M., González R., Medina A., Avaca M.S., Ginsberg S. & Aliotta S. (2012) Short term impact of artisanal dredges in a Patagonian mussel fishery: Comparisons with commercial diving and control sites. *Marine Environmental Research,* 73, 53–61.

6.27. Use alternative means of getting mussel seeds rather than dredging from natural mussel beds

- We found no studies that evaluated the effects of using alternative means of getting mussel seeds rather than dredging from natural mussel beds on subtidal benthic invertebrate 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

Mussel seeds (young mussels) used in aquaculture are often collected from areas of the seabed where the mussels naturally occur, using dredges. This can be a damaging harvest method, leading to mussel depletion and other negative impacts on other invertebrate species associated with mussel beds due to the impact from the dredge. Alternative means of collecting mussel seeds exist, such as using artificial collectors, or producing seeds in hatchery facilities, rather than dredging from natural beds (Fuentes *et al.* 1998; Maguire *et al.* 2008). Using such alternative collection methods can potentially help reduce dredging pressure and associated threats to subtidal benthic invertebrates.

Fuentes J., Molares J. & Villalba A. (1998) Growth, mortality and parasitization of mussels cultivated in the Ría de Arousa (NW Spain) from two sources of seed: intertidal rocky shore vs. collector ropes. *Aquaculture*, 162, 231–240.
Maguire J.A., Knights A.M., O'Toole M., Burnell G., Crowe T.P., Ferns M., McDonough N., McQuaid N., O'Connor B., Doyle R. & Newell C. (2008) Management recommendations for sustainable exploitation of mussel seed in the Irish Sea. *Marine Environment and Health Series*, 31.

6.28. Use an otter trawl instead of a dredge

- **One study** examined the effects of using an otter trawl instead of a dredge on subtidal benthic invertebrates. The study was in the Irish Sea[1] (Isle of Man).

COMMUNITY RESPONSE (1 STUDY)

- **Unwanted catch overall composition (1 study):** One replicated, controlled, study in the Irish Sea[1] found that an otter trawl caught a different species composition of unwanted invertebrate and fish species (combined) compared to two scallop dredges.

POPULATION RESPONSE (1 STUDY)

- **Overall abundance (1 study):** One replicated, controlled, study in the Irish Sea[1] found no difference in total invertebrate abundance and biomass living in or on the sediment of the trawl tracks following fishing with either an otter trawl or two scallop dredges.
- **Unwanted catch overall abundance (1 study):** One replicated, controlled, study in the Irish Sea[1] found that an otter trawl caught fewer unwanted invertebrates and fish (combined) compared to two scallop dredges.

OTHER (1 STUDY)

- **Commercial catch abundance (1 study):** One replicated, controlled, study in the Irish Sea[1] found that an otter trawl caught similar number of commercially targeted queen scallops compared to two scallop dredges.

Background

Dredging, for instance for bivalves, normally involves towing a heavy steel frame along the seabed, which negatively impacts subtidal benthic invertebrates due to direct physical damage and the retention of unwanted invertebrate catch, and indirectly by changes to the seabed structure and topography, such as creating dredge tracks (Currie & Parry 1996). Otter trawls have a pair of boards or metal plates (otter boards) which attach to the sides of the net and keep the net open as it is pulled through the water (Schwinghamer *et al.* 1998). Although otter boards can damage the seabed, they may cause less damage than a dredge due to the limited surface area in contact with the seabed in comparison to a dredge. Using an otter trawl instead of a dredge may potentially lessen the negative impact on subtidal benthic invertebrates (Hinz *et al.* 2012). Evidence for using other fishing gear instead of an otter trawl is summarised under "Threat: Biological resource use – Use an otter trawl instead of a beam trawl". Evidence for other interventions related to using different fishing gear is summarised under "Threat: Biological resource use".

Currie D.R. & Parry G.D. (1996) Effects of scallop dredging on a soft sediment community: a large-scale experimental study. *Marine Ecology Progress Series*, 134, 131–150.

Hinz H., Murray L.G., Malcolm F.R. & Kaiser M.J. (2012) The environmental impacts of three different queen scallop (*Aequipecten opercularis*) fishing gears. *Marine Environmental Research,* 73, 85–95.

Schwinghamer P., Gordon Jr D.C., Rowell T.W., Prena J., McKeown D.L., Sonnichsen G. & Guigné J.Y. (1998) Effects of experimental otter trawling on surficial sediment properties of a sandy-bottom ecosystem on the Grand Banks of Newfoundland. *Conservation Biology*, 12, 1215–1222.

A replicated, controlled, study (date of study not reported) in a sandy area in the north Irish Sea, Isle of Man (1) found that an otter trawl caught fewer unwanted invertebrates and fish (combined), and a different unwanted catch species composition, compared to two dredge designs. The otter trawls caught fewer unwanted invertebrates and fish (4 individuals/1,000 m^2) than the two dredge types (23–59 individuals/1,000 m^2). In addition, overall unwanted catch species composition was different between the otter trawl and the two dredges (species composition data presented as graphical analyses). Unwanted otter trawl catch was reported to be dominated by fish, whereas unwanted dredge catch was dominated by invertebrates. Following fishing with either gear, there were no changes in total invertebrate abundance and biomass living in or on the sediments (raw data not presented). The otter trawl caught similar number of commercially targeted queen scallops *Aequipecten opercularis* (45 scallops/1,000 m^2) compared to the dredges (15–48 scallops/1,000 m^2). Three queen scallop fishing gears were compared: an otter trawl, a new dredge design, and a traditional Newhaven dredge. The study site was subdivided into 12 trawling lanes (40 m wide, 1 nm long) in 20–23 m water depth. Each fishing lane was allocated to one gear design (4 lanes/design). Commercial and unwanted catches were sorted, identified, counted and weighed. Before, and seven days after fishing trials, invertebrates (size unspecified) were sampled in each lane using a 2-m beam trawl (5-min tow; 6 tows/lane) and a sediment grab (0.1 m^2; 6 grabs/lane).

(1) Hinz H., Murray L.G., Malcolm F.R. & Kaiser M.J. (2012) The environmental impacts of three different queen scallop (*Aequipecten opercularis*) fishing gears. *Marine Environmental Research,* 73, 85–95.

6.29. Use more than one net on otter trawls

- We found no studies that evaluated the effects of using more than one net on otter trawls on subtidal benthic invertebrate 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

Trawling is a method of fishing that involves pulling a fishing net (trawl) through the water behind one or more boats. Otter trawls have a pair of boards or metal plates (otter boards) which attach to the sides of the net and keep the net open as it is pulled through the water. More than one trawl can be towed simultaneously from the same boat. Towing three trawls behind one boat has been found to retain a lower weight of unwanted fish compared to a single rig in a river in Australia (Broadhurst *et al.* 2013a). Single and triple rigs have fewer otter boards than double or quad rigs and, therefore, less contact with the seabed, which may have less impact on subtidal benthic invertebrates (Broadhurst *et al.* 2013a). A study in that same Australian river found that there was no difference in the numbers or weight of unwanted fish caught between a double rig (with four otter boards) and a dual rig (with two otter boards) (Broadhurst *et al.* 2013b).

Evidence for other interventions related to otter trawl is summarised under "Threat: Biological resource use – Use an otter trawl instead of a beam trawl", and "Use an otter trawl instead of a dredge". Evidence for other interventions related to using different fishing gear is summarised under "Threat: Biological resource use".

Broadhurst M.K., Sterling D.J. & Millar R.B. (2013a) Progressing more environmentally benign penaeid-trawling systems by comparing Australian single- and multi-net configurations. *Fisheries Research,* 146, 7–17.

Broadhurst M.K., Sterling D.J. & Millar R.B. (2013b) Relative engineering and catching performances of paired penaeid-trawling systems. *Fisheries Research,* 143, 143–152.

6.30. Use an otter trawl instead of a beam trawl

- **One study** examined the effects of using an otter trawl instead of a beam trawl on subtidal benthic invertebrates. The study was in the North Sea[1] (Germany and Netherlands).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Overall abundance (1 study):** One replicated, paired, controlled study in the North Sea[1] found that otter trawls caused similar mortality of invertebrates in the trawl tracks compared to beam trawls in sandy areas but lower mortality in silty areas.

Background

Trawling is a method of fishing that involves pulling a fishing net (trawl) through the water behind one or more boats. A beam trawl is a type of trawl where the mouth of the net is held open by a wooden or metal beam, which can be up to 14 m long. Beam trawls can negatively impact subtidal benthic invertebrates through direct physical damage, bycatch, and alterations to the seabed (Bergman & Van Santbrink 2000). Other types of fishing methods may be less damaging to the seabed and its invertebrates. Otter trawls, for instance, have a pair of boards or metal plates (otter boards) which attach to the sides of the net and keep the net open as it is pulled through the water (Schwinghamer *et al.* 1998). Otter trawls are alternative fishing methods which may potentially cause less damage to the seabed and benthic invertebrates (Broadhurst *et al.* 2012).

Evidence for other interventions related to otter trawl is summarised under "Threat: Biological resource use – Use more than one net on otter trawls", and "Use an otter trawl instead of a dredge". Evidence for other interventions related to using different fishing gear is summarised under "Threat: Biological resource use".

Bergman M.J.N. & Van Santbrink J.W. (2000) Mortality in megafaunal benthic populations caused by trawl fisheries on the Dutch continental shelf in the North Sea in 1994. *ICES Journal of Marine Science*, 57, 1321–1331.

Broadhurst M.K., Sterling D.J. & Cullis B.R. (2012) Effects of otter boards on catches of an Australian penaeid trawl. *Fisheries Research,* 131–133, 67–75.

Schwinghamer P., Gordon Jr D.C., Rowell T.W., Prena J., McKeown D.L., Sonnichsen G. & Guigné J.Y. (1998) Effects of experimental otter trawling on surficial sediment properties of a sandy-bottom ecosystem on the Grand Banks of Newfoundland. *Conservation Biology*, 12, 1215–1222.

A replicated, paired, controlled study in 1992–1995 in four areas of sandy or silty seabed in the south-eastern North Sea, Netherlands and Germany (1) found that the effects of otter trawls compared to beam trawls on invertebrate mortality varied with the sediment type. Otter trawls caused similar mortality of invertebrates in the trawl tracks compared to beam trawls in sandy areas (otter: 0–41%: beam: 1–53%) but lower mortality in silty areas (otter: 1–65%: beam: 2–82%). In spring-summer 1992–1995 parallel strips (2,000 x 60 m, 300 m apart, number unspecified) in one sandy location and three silty locations were fished with either a commercially used beam trawl with tickler chains or an otter trawl. Prior to trawling, mega-invertebrates (>1 cm) and macro-invertebrates (> 1 mm) were counted from samples taken in each strip using a dredge and a sediment grab. After 24–48 h following trawling, all strips were sampled again using the same methods. Mortality (from trawling) of invertebrates present in the trawl tracks was calculated using the difference between the before and after-trawling abundances (assuming all animals killed by trawling had been eaten by predators).

(1) Bergman M.J.N. & Van Santbrink J.W. (2000) Mortality in megafaunal benthic populations caused by trawl fisheries on the Dutch continental shelf in the North Sea in 1994. *ICES Journal of Marine Science,* 57**,** 1321–1331.

6.31. Use a pulse trawl instead of a beam trawl

- **One study** examined the effects of using a pulse trawl instead of a beam trawl on subtidal benthic invertebrates. The study was in the North Sea[1] (Netherlands).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Unwanted catch abundance (1 study):** One replicated, controlled, study in the North Sea[1] found that pulse trawls caught less unwanted invertebrate catch compared to traditional beam trawls, but the effects varied with species.

OTHER (1 STUDY)

- **Commercial catch abundance (1 study):** One replicated, controlled, study in the North Sea[1] found that pulse trawls reduced the volume of commercial catch by 19% compared to beam trawls.

Background

Trawling is a method of fishing that involves pulling a fishing net (trawl) through the water behind one or more boats. A beam trawl is a type of trawl where the mouth of the net is held open by a wooden or metal beam, which can be up to 14 m long. Beam trawls can negatively impact subtidal benthic invertebrates through direct physical damage, bycatch, and alterations to the seabed (Bergman & Van Santbrink 2000). Other types of fishing methods may be less damaging to the seabed and its invertebrates. Pulse trawling is an adaptation of beam trawling which replaces tickler chains (metal chains which drag along the seabed in front of the net) with electrical drag wires that sends electric pulses into the seabed. Pulse trawls are alternative fishing methods which may potentially cause less damage to the seabed and benthic invertebrates (Despestele *et al*. 2018; Van Marlen *et al.* 2014). However, it should be noted that the use of pulse trawls is banned in many fisheries. Evidence for other interventions related to using different fishing gear is summarised under "Threat: Biological resource use".

Bergman M.J.N. & Van Santbrink J.W. (2000) Mortality in megafaunal benthic populations caused by trawl fisheries on the Dutch continental shelf in the North Sea in 1994. *ICES Journal of Marine Science,* 57**,** 1321–1331.

Depestele J., Degrendele K., Esmaeili M., Ivanović A., Kröger S., O'neill F.G., Parker R., Polet H., Roche M., Teal, L.R. & Vanelslander B. (2018) Comparison of mechanical disturbance in soft sediments due to tickler-chain SumWing trawl vs. electro-fitted PulseWing trawl. *ICES Journal of Marine Science*, 76, 312–329.

Van Marlen B., Wiegerinck J.A.M., van Os-Koomen E. & van Barneveld E. (2014) Catch comparison of flatfish pulse trawls and a tickler chain beam trawl. *Fisheries Research*, 151, 57–69.

A replicated, controlled study in 2011 in sandy areas in the North Sea, Netherlands (1) found that pulse trawls caught fewer unwanted invertebrates compared to traditional beam trawls, but the effects varied with species. Fewer unwanted invertebrates were caught when using pulse trawls compared to using beam trawls (pulse: 142 vs beam: 177

individuals/ha). However, when sorted by groups, pulse trawls caught fewer invertebrates living on the sediments (131 vs 175) but more living inside the sediment (11 vs 2), compared to beam trawls. In particular, pulse trawls caught fewer echinoderms (82 vs 113) and gastropods (sea snails; 0.0 vs 0.1), compared to the beam trawl, similar numbers of anthozoan (0.0 vs 0.1), bivalves (0.1 vs 0.2), cephalopods (0.1 vs 0.2), and crustaceans (60 vs 64). Pulse trawls also caught 57% less total discards (non-commercial unwanted catch of invertebrates and fish) by volume (0.25 vs 0.29 basket/ha). The pulse trawl reduced the volume of commercial catch by 19% compared to the traditional trawl (0.08 vs 0.1 basket/ha). Pulse (electrical) trawling was prohibited in European fisheries in 1998, but a system of derogations set up in 2006 has allowed the practice to continue, including experimental trials. Comparison trials were conducted in May 2011 with three vessels fishing side-by-side (two boats using pulse trawls, one using traditional flat-fish tickler chain beam trawls). Catches from 33 trawls/vessel were assessed. The total discard volume was measured. Invertebrate discards were identified and counted from one subsample of total catch/trawl (35 kg basket). As of 2019, the practice has been fully banned in European waters.

(1) Van Marlen B., Wiegerinck J.A.M., van Os-Koomen E. & van Barneveld E. (2014) Catch comparison of flatfish pulse trawls and a tickler chain beam trawl. *Fisheries Research*, 151, 57–69.

6.32. Use a smaller beam trawl

- **One study** examined the effects of using a smaller beam trawl on subtidal benthic invertebrates. The study was in the North Sea[1] (Germany and Netherlands).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Overall abundance (1 study):** One replicated, paired, controlled study in the North Sea[1] found that a smaller beam trawl caused similar mortality of invertebrates in the trawl tracks compared to a larger beam trawl.

Background

Trawling is a method of fishing that involves pulling a fishing net (trawl) through the water behind one or more boats. A beam trawl is a type of trawl where the mouth of the net is held open by a wooden or metal beam, which can be up to 14 m long. Beam trawls can negatively impact subtidal benthic invertebrates through direct physical damage, bycatch, and alterations to the seabed (Bergman & Van Santbrink 2000). A smaller beam could be used, which may potentially limit impact on subtidal benthic invertebrates through reduced damage to the seabed. Evidence related to the use of other fishing gear instead of a beam trawl is summarised under "Threat: Biological resource use – Use a pulse trawl instead of a beam trawl" and "Use an otter trawl instead of a beam trawl".

Bergman M.J.N. & Van Santbrink J.W. (2000) Mortality in megafaunal benthic populations caused by trawl fisheries on the Dutch continental shelf in the North Sea in 1994. *ICES Journal of Marine Science*, 57, 1321–1331.

A replicated, paired, controlled study in 1992-1995 in four areas of silty or sandy seabed in the south-eastern North Sea, Netherlands and Germany (1) found that using a smaller beam trawl caused similar mortality of invertebrates in the trawl tracks compared to using a larger beam trawl. Mortality using a 4-m beam trawl varied between 2 to 80% depending on species, similar to a 12-m beam trawl (1–82% mortality). Mortality did not differ across sediment type (sandy or silty). In spring-summer 1992–1995, parallel strips (2,000 x 60 m, 300 m apart, number unspecified) in one sandy location and three silty locations were fished with either a 12-m (commercially used) or 4-m beam trawl (both with tickler chains). Prior to trawling, mega-invertebrates (>1 cm) and macro-invertebrates (>1 mm) were counted from samples taken from each strip using a dredge and a sediment grab. After 24–48 h following trawling, all strips were sampled again using the same methods. Mortality (from trawling) of invertebrates present in the trawl tracks was calculated using the difference between the before and after-trawling abundances (assuming all animals killed by trawling had been eaten by predators).

(1) Bergman M.J.N. & Van Santbrink J.W. (2000) Mortality in megafaunal benthic populations caused by trawl fisheries on the Dutch continental shelf in the North Sea in 1994. *ICES Journal of Marine Science,* 57**,** 1321–1331.

6.33. Modify trawl doors to reduce sediment penetration

- We found no studies that evaluated the effects of modifying trawl doors to reduce sediment penetration on subtidal benthic invertebrate 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

Trawling is a method of fishing that involves pulling a fishing net (trawl) through the water behind one or more boats. Trawl doors are the boards or metal plates (otter boards) which attach to the sides of the net and keep the net open as it is pulled through the water. The trawl doors can be modified, for instance by using smaller otter boards, or securing netting at wing ends, to reduce sediment penetration. This may potentially limit impact on subtidal benthic invertebrates through reduced damage to the seabed (Balash *et al.* 2016).

Balash C., Sterling D. & Broadhurst M.K. (2016) Progressively evaluating a penaeid W trawl to improve eco-efficiency. *Fisheries Research*, 181, 148–154.

6.34. Outfit trawls with a raised footrope

- We found no studies that evaluated the effects of outfitting trawls with a raised footrope on subtidal benthic invertebrate 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

Trawling is a method of fishing that involves pulling a fishing net (trawl) through the water behind one or more boats. The footrope consists of a rope, wire or chain which is attached to the bottom front of the net (the lower edge of the net mouth) to provide weight to keep the net on or near the seabed. Footrope configuration varies with trawls and the commercial species targeted, and can affect the level of negative impacts on the seabed and subtidal benthic invertebrates (Hannah *et al.* 2013). To potentially reduce contact with the seabed, and therefore direct damage and disturbance, the footrope can be raised.

Hannah R.W., Lomeli M.J. & Jones S.A. (2013) Direct estimation of disturbance rates of benthic macroinvertebrates from contact with standard and modified ocean shrimp (*Pandalus jordani*) trawl footropes. *Journal of Shellfish Research*, 32, 551–558.

6.35. Limit the maximum weight and/or size of bobbins on the footrope

- We found no studies that evaluated the effects of limiting the maximum weight and/or size of bobbins on the footrope on subtidal benthic invertebrate 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

The footrope consists of a rope, wire or chain which is attached to the bottom front of the net (the lower edge of the net mouth) to provide weight. Bobbins, rollers or other hard material encircle or are tied along the footrope to bounce or pivot over seabed obstructions preventing the footrope and net from snagging on the seabed. Footrope configuration varies with trawls and the commercial species targeted, and can affect the level of negative impacts on the seabed and subtidal benthic invertebrates (Hannah *et al.* 2013). Large, heavy bobbins and rollers can damage the seabed, therefore affecting subtidal benthic invertebrates. Setting limits on their weight and/or size can potentially reduce damage to the seabed and associated impacts on subtidal benthic invertebrates.

Hannah R.W., Lomeli M.J. & Jones S.A. (2013) Direct estimation of disturbance rates of benthic macroinvertebrates from contact with standard and modified ocean shrimp (*Pandalus jordani*) trawl footropes. *Journal of Shellfish Research*, 32, 551–558.

6.36. Fit a funnel (such as a sievenet) or other escape devices on shrimp/prawn trawl nets

- **One study** examined the effects of fitting a funnel, sievenet, or other escape devices on trawl nets on marine subtidal invertebrate. The study was in the North Sea[1] (UK).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Unwanted catch abundance (1 study):** One replicated, paired, controlled study in the North Sea[1] found that trawl nets fitted with a sievenet appeared to catch fewer unwanted catch of non-commercial invertebrates compared to unmodified nets.

Background

Trawling is a method of fishing that involves pulling a cone-shaped fishing net (trawl) through the water behind one or more boats. The net is wide at the opening and narrows to a bag or 'codend', tied at the end with a drawstring, where organisms are trapped. Trawl nets can catch a considerable number of unwanted organisms, including non-commercially targeted species and organisms under the legal-size limit. To reduce the amount of unwanted organisms, a net can be modified by inserting a funnel-like device (such as a sievenet) before the codend (Santos *et al.* 2018). This device is designed to direct unwanted catch to an escape hole in the body of the trawl. The idea is that the target species go over the hole in the net, while non-target species can escape through the release hole. These funnel-like devices are usually not made of rigid material and therefore can be more acceptable to fishers than a rigid sorting grid (evidence summarised under "Threat: Biological resource use – Fit one or more soft, semi-rigid, or rigid grids or frames to trawl nets").

Santos J., Herrmann B., Mieske B., Krag L. A., Haase S. & Stepputtis D. (2018) The efficiency of sieve-panels for bycatch separation in *Nephrops* trawls. *Fisheries Management and Ecology*, 25, 464–473.

A replicated, paired, controlled study in 2006–2007 in the North Sea, off the east coast of England, UK (1) found that trawl nets used in shrimp/prawn fisheries fitted with a sievenet (funnel-like device) appeared to catch fewer unwanted non-commercial invertebrates (discard) compared to unmodified nets. Differences were not statistically tested. Of the seven invertebrate discard species recorded, six tended to be caught in lower numbers in nets fitted with a sievenet compared to nets without (28–83% reduction in numbers caught), and one species tended to be caught in equal numbers. Use of selective gear to reduce unwanted catch in the brown shrimp fishery was made compulsory in 2003 in the European Union. Between January 2006 and January 2007, abundances of unwanted invertebrate species were compared in nets with a sievenet and without. Nets were deployed in pairs (one sievenet; one unmodified net) during 98 hauls for 1h. All organisms were identified, sorted as commercial catch or discard, and counted.

(1) Catchpole T.L., Revill A.S., Innes J. & Pascoe S. (2008) Evaluating the efficacy of technical measures: a case study of selection device legislation in the UK *Crangon crangon* (brown shrimp) fishery. *ICES Journal of Marine Science*, 65, 267–275.

6.37. Fit one or more mesh escape panels/windows to trawl nets

- **Seven studies** examined the effects of adding one or more mesh escape panels/windows to trawl nets on subtidal benthic invertebrate populations. Six were in the North Sea[1a–b,2,3a–b,5] (Belgium, Netherlands, UK), two in the Thames estuary[1a–b] (UK), one in the English Channel[2] (UK), and one in the Gulf of Carpentaria[4] (Australia).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (7 STUDIES)

- **Overall survival (1 study):** One replicated, paired, controlled study in the English Channel and the North Sea[2] found that fitting nets with either one of seven designs of square mesh escape panels (varying mesh size and twine type) led to higher survival rates of invertebrates that escaped the nets compared to unmodified nets.
- **Unwanted catch overall abundance (7 studies):** Three of seven replicated, paired, controlled studies in the North Sea[1a–b,2,3a–b,5], the Thames estuary[1a–b], the English Channel[2] and the Gulf of Carpentaria[4] found that trawl nets fitted with one or more mesh escape panels/windows/zones

reduced the unwanted catch of invertebrates compared to unmodified nets[1b,2,5]. Two found mixed effects of fitting escape panels on the unwanted catch of invertebrates and fish[3a–b] depending on the panel design. Two found that trawl nets fitted with escape panels[1a,4] caught similar amounts of unwanted invertebrates and fish[4] compared to unmodified nets.

OTHERS (7 STUDIES)

- **Commercially targeted catch abundance (7 studies):** Three of seven replicated, paired, controlled studies in the North Sea[1a–b,2,3a–b,5], the Thames estuary[1a–b], the English Channel[2] and the Gulf of Carpentaria[4], found that trawl nets fitted with one or more mesh escape panels/windows/zones caught similar amounts of all or most commercial species to unmodified nets[1a,3a–b]. Three found mixed effects of fitting escape panels on the catch of all or most commercial species depending on the species and/or panel design[1b,2,4]. One found that trawl nets fitted with escape panels reduced the catch of commercial species[5] compared to unmodified nets.

Background

Trawling is a method of fishing that involves pulling a cone-shaped fishing net (trawl) through the water behind one or more boats. The net is wide at the opening and narrows to a bag or 'codend', tied at the end with a drawstring, where organisms are trapped. Trawl nets can catch a considerable number of unwanted organisms, including non-commercially targeted species and organisms under the legal-size limit. Standard trawl nets are made from diamond-shaped mesh. To potentially reduce the amount of unwanted organisms, a net can be modified by fitting one or more mesh "escape panels" in the outer mesh of the net before the codend (Brčić *et al*. 2017; Fonteyne & Polet 2002; Revill & Jennings 2005). These panels are sections of netting made from a different mesh design than the rest of the net, for instance made from square mesh or large diamond mesh. The panels are designed to allow smaller unwanted organisms to escape, while retaining the commercially targeted ones. Mesh panels, which are also referred to as escape windows, escape zones, and drop-out panels, or more generally as "bycatch reduction devices", can be fitted directly before the codend, or elsewhere on the main body of the net such as behind the groundrope. Here, we included escape zones such as "Bigeye" and "Fisheye" devices, as although they do not strictly use mesh sections, they have similar functions of letting organisms escape through a specific area of the net.

Brčić J., Herrmann B. & Sala A. (2017) Can a square-mesh panel inserted in front of the cod end improve size and species selectivity in Mediterranean trawl fisheries? *Canadian Journal of Fisheries and Aquatic Sciences*, 75, 704–713.

Fonteyne R. & Polet H. (2002) Reducing the benthos by-catch in flatfish beam trawling by means of technical modifications. *Fisheries Research*, 55, 219–230.

Revill A.S. & Jennings S. (2005) The capacity of benthos release panels to reduce the impacts of beam trawls on benthic communities. *Fisheries Research*, 75, 73–85.

A replicated, paired, controlled study in 1999 in two soft seabed areas in the southern North Sea, Belgium and Thames estuary, UK (1a) found that a modified trawl net with either diamond or square mesh escape zones ("bycatch reduction device") did not reduce the amount of unwanted invertebrate catch overall compared to a standard unmodified net, and had mixed effects on the catch of individual species. The overall weight of unwanted invertebrates caught was not significantly different from standard nets for the nets with diamond mesh escape zones (diamond: 91; standard: 103 kg) or square mesh escape zones (square: 137; standard: 142 kg). Of the 11 unwanted invertebrate species caught, nets with diamond mesh escape zones reduced the catch of one, increased the catch of two, and caught similar amounts of the remaining eight, compared to standard nets. Nets with square mesh escape zones caught similar amounts of all species. Both designs of escape zones caught similar amounts for five of the six commercial species caught and reduced the catch of one species (plaice *Pleuronectes platessa*) by 15–18%. The escape zone (large diamond or large square mesh; 400 mm) were fitted to a beam trawl net (4 m) behind the ground rope. Fishing took place simultaneously with one modified and one standard unmodified net by attaching the two nets to an 8 m beam. Hauls (10 for diamond mesh; 6 for square mesh) were conducted in March 1999 in 20–50 m water depth. Total catch weights were recorded, and all invertebrate species were separated, weighed and identified to species level.

A replicated, controlled study in 1999–2000 in two soft seabed areas in the southern North Sea, Belgium and Thames estuary, UK (1b) found that overall when fitted to trawl nets, square mesh windows ("bycatch reduction device") of three different sizes reduced unwanted catch of invertebrates, compared to nets without a device, and had mixed effects on the catch of individual species. The windows decreased the overall weight of unwanted invertebrates caught by 64–83% compared to unmodified nets. The 120 mm window significantly decreased catches of six of 16 species (45–95% reduction). The 150 mm window significantly decreased catches of 11 of 17 species (34–90% reduction). The 200 mm window significantly decreased catches of four of 16 species (92–97% reduction). The 120 mm window increased the catch of one of eight commercially targeted species (by 111%) compared to nets without a device, with no differences for the remaining seven. The 150 mm window did not impact the catch of any commercially targeted species. The 200 mm window decreased catches of two of eight commercially targeted species (by 23–45%) compared to nets without a device, with no differences for the remaining six. Windows of either 120 mm, 150 mm or 200 mm mesh size were fitted to a beam trawl net (4 m) just in front of the codend. Nets were deployed between November and February 2000 at 20–50 m depth during paired hauls (one net with and one without a device; 5–16 hauls/window type; by attaching the two nets to one 8 m beam). All unwanted invertebrates were identified, counted, and weighed. Commercial catches were identified and weighed. No comparisons were made between windows of different mesh sizes.

A replicated, paired, controlled study in 2002–2004 in six seabed areas in the western English Channel and the North Sea, UK, Denmark, Germany, Netherlands (2) found that nets fitted with either one of seven designs of square mesh benthos-release panels ("bycatch reduction device") caught less non-commercial unwanted catch of invertebrates (discard), compared to unmodified nets, and invertebrates escaping the nets had high survival rates. The two designs that reduced discards the most compared to unmodified nets consisted of 150 mm mesh with 5 mm diameter double twine (with panel: 1,988 individuals caught, without: 9,802 individuals) and 150 mm mesh with 6 mm diameter single twine (with panel: 5,286 individuals, without: 21,128 individuals). Overall survival rates (all designs combined) of escaped invertebrates were high (93–100% depending on species). In addition, five of the seven designs caught a similar amount of commercially targeted species (including the two that led to the greatest reductions in discards). These five designs reduced invertebrate discard by 48–80%. The other two designs led to 17–20% losses of target species (reductions in invertebrate discards not shown). The designs were tested on commercial beam trawls at 20–80 m depth. One trawl fitted with a panel and an unmodified trawl were towed simultaneously

(4–24 tows/design). All commercial fish caught during the trials were counted and measured, and benthic invertebrates were counted and identified to species level. Invertebrates that had escaped through the panels were caught in a sled fitted to the underside of the trawl, and their survival in tanks assessed over three days.

A replicated, paired, controlled study in 1999 in one seabed area in the North Sea, Netherlands (3a) found that trawl nets modified by adding one of two designs of diamond mesh drop out panels ("bycatch reduction device") caught less non-commercial unwanted species of invertebrates and fish (combined as discards) compared to unmodified trawl nets. Nets fitted with a 720 mm mesh panel caught less discard (75 kg/h) than unmodified nets (87 kg/h), but nets fitted with a 120 mm mesh panel caught similar amounts (33 kg/h) to unmodified nets (34 kg/h). All modified nets caught similar amounts of commercial species (14–17 kg/h) to unmodified nets (14–15 kg/h). In January 1999, a trawl net fitted with a panel design (escape zone; each panel had 19 diamonds of either 720 mm or 120 mm) was compared to an unmodified net during 14–18 paired simultaneous deployments along parallel strips (2,000 x 30 m). Catches were sorted into commercial species (fishery target and other commercially valuable species) and discards, and each group weighed.

A replicated, paired, controlled study in 1999 in one seabed area in the North Sea, Netherlands (3b) found that for three of four panel designs, trawl nets modified by adding a diamond mesh drop out panel ("bycatch reduction device") reduced the amount of non-commercial unwanted species of invertebrates and fish (combined as discards) compared to unmodified trawl nets. Nets fitted with either one of three drop out panel designs caught less discards (94–110 kg/h) than unmodified nets (123–128 kg/h). Nets fitted with the fourth design (16 meshes of 100 mm) caught similar amounts (102 kg/h) to unmodified nets (136 kg/h). All modified nets caught similar amounts of commercial species (14–17 kg/h) to unmodified nets (14–15 kg/h). Four panel designs (escape zone) were tested on trawl nets: 19 diamonds of 500 mm; 19 diamonds of 100 mm; 16 diamonds of 100 mm; 12 diamonds of 100 mm. In March 1999, each design was compared to an unmodified net during 5–12 paired simultaneous deployments along parallel strips (2,000 x 30 m). Catches were sorted into commercial species (fishery target and other commercially valuable species) and discards, and each group weighed.

A replicated, paired, controlled study in 2001 in seabed areas in the Gulf of Carpentaria, northern Australia (4) found that nets fitted with either one of two escape zone design ("bycatch reduction device") did not reduce the numbers of large sponges caught or weight of small unwanted catch (invertebrates and fish combined), compared to unmodified nets. Data were not provided. Nets fitted with a 'Bigeye' escape zone reduced the catch of commercially targeted prawns by 4.2% compared to an unmodified net, while nets fitted with a square-mesh escape panel caught similar amounts. The use of a "bycatch reduction device" has been compulsory since 2000 in the Australian prawn fishery (as well as the use of a "turtle excluder device"). Commercial vessels towed twin Florida Flyer prawn trawl nets from each side of the vessel in August–November 2001. Nets fitted with one of the two designs of escape zone (112 nets examined for small bycatch, 97 for sponges) and an unmodified net (703 for small bycatch, 339 for sponges) were randomly assigned to either side of the vessel. Total weights of small unwanted catch (<300 mm), commercially targeted prawns, and counts of sponges (>300 mm) were recorded. The "Bigeye" design was later removed from the Australian list of approved designs.

A replicated, paired, controlled study in 2014–2015 in five seabed areas in the North Sea, UK and Belgium (5) found that overall, when fitted to trawl nets, all four designs of square-meshed window ("bycatch reduction device") tested reduced the non-commercial unwanted catch of invertebrates (discard), compared to unmodified nets without a device. Fitting nets with either design (window of either 150 mm, 200 mm, 240 mm mesh, or a 240 mm window with electrical pulse; see paper for details) decreased the catch of all invertebrate discard species recorded by 9–100% compared to unmodified nets. The 150 mm window significantly decreased catches of 11 of 15 species (55–91% reduction). The 200 mm window significantly decreased catches of 10 of 14 species (9–92% reduction). The 240 mm window significantly decreased catches of nine of 18 species (38–97% reduction). The electrified 240 mm window significantly decreased catches of 15 of 19 species (58–100% reduction). All devices reduced catches of commercially targeted species compared to nets without a device, by between 5 and 22%. Invertebrate discard was compared in nets with and without a device. Nets were deployed by two vessels during 58 paired hauls for 1.5 hour (one net with and one without a device; 10–22 hauls/device type). All invertebrate discards were identified, counted, and weighed from 5–8 kg subsamples. Commercially targeted catches were weighed. No comparisons were made between windows of different designs.

(1a–b) Fonteyne, R. & Polet H. (2002) Reducing the benthos by-catch in flatfish beam trawling by means of technical modifications. *Fisheries Research,* 55, 219–230.
(2) Revill A.S. & Jennings S. (2005) The capacity of benthos release panels to reduce the impacts of beam trawls on benthic communities. *Fisheries Research,* 75, 73–85.
(3a–b) Van Marlen B., Bergman M.J.N., Groenewold S. & Fonds M. (2005) New approaches to the reduction of non-target mortality in beam trawling. *Fisheries Research,* 72, 333–345.
(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) Soetaert M., Lenoir H. & Verschueren B. (2016) Reducing bycatch in beam trawls and electrotrawls with (electrified) benthos release panels. *ICES Journal of Marine Science*, 73, 2370–2379.

6.38. Fit one or more soft, semi-rigid, or rigid grids or frames to trawl nets

- **Two studies** examined the effects of fitting one or more soft, semi-rigid, or rigid grids or frames to trawl nets on subtidal benthic invertebrate populations. The studies were in the Gulf of Carpentaria[1] and Spencer Gulf[2] (Australia).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Unwanted catch abundance (1 study):** Two replicated, paired, controlled studies in the Gulf of Carpentaria[1] and in Spencer Gulf[2] found that nets fitted with a 'downward'-oriented grid but not 'upward'-oriented grid reduced the weight of small unwanted catch and that both grid orientations caught fewer unwanted large sponges[1], and that nets fitted with two sizes of grids reduced the number and biomass of unwanted blue swimmer crabs and giant cuttlefish caught[2], compared to unmodified nets.

OTHER (2 STUDIES)

- **Commercial catch abundance (2 studies):** Two replicated, paired, controlled studies in the Gulf of Carpentaria[1] and Spencer Gulf[2] found that nets fitted with a 'downward'-oriented grid[1] or a small grid[2] reduced the catch of commercially targeted prawns, compared to unmodified nets, but those fitted with an 'upward'-oriented grid[1] or a large grid[2] caught similar amounts to unmodified nets.

Background

Trawling is a method of fishing that involves pulling a cone-shaped fishing net (trawl) through the water behind one or more boats. The net is wide at the opening and narrows to a bag or 'codend', tied at the end with a drawstring, where organisms are trapped. Trawl nets can catch a considerable number of unwanted organisms, including non-commercially targeted species and organisms under the legal-size limit. To reduce the amount of unwanted organisms, one or more soft, semi-rigid or rigid grids or frames can be fitted to the inner side of the net before the codend. This grid or frame is designed to prevent larger organisms, such as turtles or sharks, from entering the net/codend, while retaining the commercially targeted organisms (Brčić *et al.* 2015; Sala *et al.* 2011).

Evidence related to the use of grids in combination with other "bycatch reducing devices" are summarised under "Threat: Biological resource use – Fit one or more mesh escape panels/windows and one or more soft, rigid or semi-rigid grids or frames to trawl nets" and in combination with a modified codend under "Fit one or more soft, semi-rigid, or rigid grids or frames to trawl nets and use square mesh instead of a diamond mesh at the codend".

Brčić J., Herrmann B., De Carlo F. & Sala A. (2015) Selective characteristics of a shark-excluding grid device in a Mediterranean trawl. *Fisheries Research*, 172, 352–360.

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.

A replicated, paired, controlled study in 2001 in areas of seabed in the Gulf of Carpentaria, northern Australia (1) found that the effects of fitting a grid ("turtle excluder device") to trawl nets, on large sponges and small unwanted catch (invertebrates and fish combined) varied with the device orientation. Nets fitted with a device oriented either 'downward' or 'upward' caught 82–96% fewer large sponges, compared to unmodified nets, but only the 'downward' devices reduced the weight of small unwanted catch (by 8%; data not provided for the 'upward' device). Compared to unmodified nets, nets fitted with a 'downward' device reduced the catch of commercially targeted prawns by 6%, while those with an 'upward' device caught similar amounts. The use of a "turtle excluder device" has been compulsory since 2000 in the Australian prawn fishery (as well as the use of a "bycatch reduction device"). Commercial vessels towed twin Florida Flyer prawn trawl nets from each side of the vessel in August–November 2001. Nets with one of 23 grid designs (rigid or semi-rigid frame with ≤120 mm bar spacing and an opening of ≥700 mm) grouped as either 'upward' (9 devices) or 'downward' (14 devices) oriented (267 nets examined for small unwanted catch, 392 for sponges) and an unmodified net (339 for sponges, 703 for small unwanted catch) were randomly assigned to either side of the vessel. Total weights of small unwanted catch (<300 mm), commercially targeted prawns, and counts of sponges (>300 mm) were recorded.

A replicated, paired, controlled study in 2014 in a sandy area in Spencer Gulf, Southern Australia (2) found that, when fitted to trawl nets, two grids reduced the number and biomass of unwanted giant cuttlefish *Sepia apama* and blue swimmer crabs *Portunus armatus* caught, compared to conventional nets without grids. Compared to conventional nets, nets fitted with a small grid resulted in a 50% decrease in the number and a 60% decrease in the biomass of giant cuttlefish caught, as well as a 40% decrease in the number and a 48% decrease in the biomass of blue swimmer crab caught. Nets fitted with a large grid resulted in 34% decrease in the number and a 37% decrease in the biomass of giant cuttlefish caught, as well as a 34% decrease in the number and a 50% decrease in the biomass of blue swimmer crab caught. There were no differences in cuttlefish abundance and biomass between the grid sizes. Catch of commercially targeted western king prawns *Melicertus latisulcatus* was reduced by 8% when using a small grid compared to a large grid and the conventional net (which had identical catches). Two grids were tested: a small grid (1.4 m long, 45° angle) and a large grid (1.98 m long, 30° angle) (see paper for full details). For 30 min at night, a trawler towed two identical nets (one on each side) fitted with a 41 mm mesh codend during simultaneous, paired deployments: one net fitted with a grid and one unmodified conventional net (eight deployments using small grids, seven using large grids). For each deployment, the weight and numbers of cuttlefish and crabs were recorded, as well as the weight of other unwanted catch. Weight and size of prawns were also recorded.

(1) 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.
(2) Kennelly S.J. & Broadhurst M.K. (2014) Mitigating the bycatch of giant cuttlefish *Sepia apama* and blue swimmer crabs *Portunus armatus* in an Australian penaeid-trawl fishery. *Endangered Species Research*, 26, 161–166.

6.39. Fit one or more mesh escape panels/windows and one or more soft, rigid or semi-rigid grids or frames to trawl nets

- **One study** examined the effects of fitting one or more mesh escape panels/windows and one or more soft, rigid or semi-rigid grids or frames to trawl nets on subtidal benthic invertebrate populations. The study was in the Gulf of Carpentaria[1] (Australia).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Unwanted catch abundance (1 study):** One replicated, paired, controlled study in Gulf of Carpentaria[1] found that trawl nets fitted with an escape window and a grid reduced the total weight of small unwanted catch and caught fewer unwanted large sponges, compared to unmodified nets.

OTHER (1 STUDY)

- **Commercial catch abundance (1 study):** One replicated, paired, controlled study in Carpentaria[1] found that trawl nets fitted with an escape window and a grid reduced the catch of commercially targeted prawns, compared to unmodified nets.

Background

Trawling is a method of fishing that involves pulling a cone-shaped fishing net (trawl) through the water behind one or more boats. The net is wide at the opening and narrows to a bag or 'codend', tied at the end with a drawstring, where organisms are trapped. Trawl nets can catch a considerable number of unwanted organisms, including non-commercially targeted species and organisms under the legal-size limit. To reduce the amount of unwanted organisms, a net can be modified by fitting one or more mesh "escape panels" in the outer mesh of the net before the codend in combination with one or more "grids" to the inner side of the net before the codend (Brewer *et al.* 2006). The grid is designed to prevent them entering the net and/or codend (Brčić *et al.* 2015) and the mesh panels are designed to allow them to escape the net (Revill & Jennings 2005). Mesh panels, which can also be referred to as escape windows, escape zones, and drop-out panels, or more generally as "bycatch reduction devices", can be fitted directly before the codend, or elsewhere on the main body of the net such as behind the groundrope. Here, we included escape zones such as "Bigeye" and "Fisheye" devices, as although they do not strictly use mesh sections, they have similar functions of letting organisms escape through a specific area of the net.

Evidence related to the use of each modification (mesh panels and grids) separately, are summarised under "Threat: Biological resource use – Fit one or more mesh escape panels/windows to trawl nets" and "Fit one or more soft, rigid or semi-rigid grids or frames to trawl nets".

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.

Brčić J., Herrmann B., De Carlo F. & Sala A. (2015) Selective characteristics of a shark-excluding grid device in a Mediterranean trawl. *Fisheries Research*, 172, 352–360.

Revill A.S. & Jennings S. (2005) The capacity of benthos release panels to reduce the impacts of beam trawls on benthic communities. *Fisheries Research*, 75, 73–85.

A replicated, paired, controlled study in 2001 in areas of seabed in the Gulf of Carpentaria, northern Australia (1) found that nets fitted with a mesh escape window ("bycatch reduction device") and a grid ("turtle excluder device"), caught fewer large sponges and reduced the total weight of small unwanted catch (invertebrates and fish combined), compared to unmodified nets. Nets fitted with both escape window and grid caught 85% fewer large sponges and reduced the weight of small unwanted catch by 8%, compared to unmodified nets. The modified nets reduced the catch of commercially targeted prawns by 6%. The use of a "turtle excluder device" and a "bycatch reduction device" has 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 both one of two designs of escape window (a "Bigeye" design or a square-mesh escape window) and one of 23 grid designs (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 (324 modified nets examined for small unwanted catch, 150 for sponges; 703 unmodified nets for small unwanted catch, 339 for sponges). Total weights of small unwanted catch (<300 mm), commercially targeted prawns, and counts of sponges (>300 mm) were recorded. The combinations of various device designs were not compared. The "Bigeye" design was later removed from the Australian list of approved escape zone designs.

(1) 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.

6.40. Use a larger codend mesh size on trawl nets

- **One study** examined the effects of using a larger codend mesh size on trawl nets on unwanted catch of subtidal benthic invertebrate populations. The study was in the Gulf of Mexico[1] (Mexico).

COMMUNITY RESPONSE (1 STUDY)

- **Unwanted catch species richness/diversity (1 study):** One replicated, paired, controlled study in the Gulf of Mexico[1] found that trawl nets fitted with a larger mesh codend caught fewer combined species of non-commercial unwanted invertebrates and fish compared to a traditional codend.

POPULATION RESPONSE (1 STUDY)

- **Unwanted catch abundance (1 study):** One replicated, paired, controlled study in the Gulf of Mexico[1] found that trawl nets fitted with a larger mesh codend caught lower combined biomass and abundance of non-commercial unwanted invertebrates and fish compared to a traditional codend.

OTHER (1 STUDY)

- **Commercial catch abundance (1 study):** One replicated, paired, controlled study in the Gulf of Mexico[1] found that trawl nets fitted with a larger mesh codend caught less biomass and abundance of commercially targeted shrimps compared to a traditional codend, but that the biomass ratios of commercially targeted to discard species was similar for both.

Background

Trawling is a method of fishing that involves pulling a fishing net (trawl) through the water behind one or more boats. Trawl nets can catch a considerable number of unwanted organisms, including non-commercially targeted species and organisms under the legal-size limit. Nets are traditionally made of diamond-shaped mesh throughout, including at the codend. To reduce the amount of unwanted organisms caught, a codend made of larger size mesh can be used, with the aim to allow smaller unwanted organisms to escape, while retaining the commercially targeted ones (Burgos-León *et al*. 2009).

Evidence related to other codend modifications are summarised under "Threat: Biological resource use – Use a square mesh instead of a diamond mesh codend on trawl nets".

Burgos-León A., Pérez-Castañeda R. & Defeo O. (2009) Discards from the artisanal shrimp fishery in a tropical coastal lagoon of Mexico: spatio-temporal patterns and fishing gear effects. *Fisheries Management and Ecology*, 16, 130–138.

A replicated, paired, controlled study (year unspecified) of three estuarine sites in the Celestun Lagoon, Gulf of Mexico, Mexico (1) found that trawl nets fitted with a 2.5 cm mesh codend instead of a traditional 1.3 cm mesh caught fewer combined non-commercial unwanted invertebrate and fish species (discard) and lower biomass and abundance of discarded organisms. On average, nets with the larger mesh codend caught fewer discard species (3–12) than nets with the traditional codend (12–20). Biomass and abundance of discards were on average lower with the larger mesh codend (biomass: 2–10 g/245 m^2; abundance: 0.7 individuals/245 m^2) than with the traditional codend (biomass: 22–57; abundance: 37). Nets with the larger mesh codend also caught less biomass and abundance of commercially targeted shrimps (biomass: 3–15 g/245 m^2; abundance: 1 individual/245 m^2) than nets with the traditional codend (biomass: 15–39; abundance: 20). This however led to similar biomass ratios of commercially targeted to discard species for both mesh sizes (1:1). On three occasions at each of three sites, a vessel towed two identical bottom-nets simultaneously over 100 m during paired deployments, one fitted with the traditional codend, the other with the larger mesh codend. For each deployment, discarded organisms were identified and their combined weight and counts recorded. Weights of commercially targeted shrimps were also recorded.

(1) Burgos-León A., Pérez-Castañeda R. & Defeo O. (2009) Discards from the artisanal shrimp fishery in a tropical coastal lagoon of Mexico: spatio-temporal patterns and fishing gear effects. *Fisheries Management and Ecology*, 16, 130–138.

6.41. Use a square mesh instead of a diamond mesh codend on trawl nets

- **One study** examined the effects of using a square mesh instead of a diamond mesh codend on trawl nets on unwanted catch of subtidal benthic invertebrate populations. The study was in the English Channel[1] (UK).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Unwanted catch abundance (1 study):** One replicated, paired, controlled study in the English Channel[1] found that a trawl net with a square mesh codend caught less non-commercial unwanted invertebrates in one of two areas, and similar amounts in the other area, compared to a standard diamond mesh codend.

OTHER (1 STUDY)

- **Commercial catch abundance (1 study):** One replicated, paired, controlled study in the English Channel[1] found that a trawl net with a square mesh codend caught similar numbers of commercially targeted fish species in two areas, and that in one of two areas it caught more commercially important shellfish, compared to a standard diamond mesh codend.

Background

Trawling is a method of fishing that involves pulling a fishing net (trawl) through the water behind one or more boats. Trawl nets can catch a considerable number of unwanted organisms, including non-commercially targeted species and organisms under the legal-size limit. Nets are traditionally made of diamond-shaped mesh throughout, including at the codend. To reduce the amount of unwanted organisms caught, a codend made of square mesh can be used instead of diamond mesh, with the aim to allow smaller unwanted organisms to escape, while retaining the commercially targeted ones (Broadhurst *et al.* 2010).

Evidence related to the use of modified codend in combination with a mesh panel ("bycatch reduction device") or a grid ("turtle excluder device") are summarised under "Threat: Biological resource use – Fit one or more mesh escape panels/windows to trawl nets and use a square mesh instead of a diamond mesh codend" and "Fit one or more soft, semi-rigid, or rigid grids or frames to trawl nets and use a codend of different mesh geometry or size". Evidence related to other codend modifications are summarised under "Threat: Biological resource use – Use a larger mesh codend".

Broadhurst M.K., Millar R.B. & Brand C.P. (2010) Diamond-vs. square-mesh codend selectivity in southeastern Australian estuarine squid trawls. *Fisheries Research*, 102, 276–285.

A replicated, paired, controlled study in 2007 in two areas of the English Channel, southwest England, UK (1) found that in one of two areas a trawl net fitted with a square mesh codend caught less non-commercial unwanted invertebrates (discard) compared to a standard trawl fitted with a diamond mesh codend. In one of two areas examined, using a square mesh codend instead of a standard diamond mesh codend reduced the biomass of invertebrate discard by 11% (square: 794 vs diamond: 889 kg). The square mesh codend caught similar numbers of commercially targeted species (megrim and anglerfish) in that area to the standard trawl and caught 26% more commercially important shellfish. In the other area, the square mesh codend caught similar biomass of invertebrate discard (1,746 kg) as the diamond mesh codend (1,842 kg), and similar numbers of commercially targeted species (Dover sole and plaice in that area). Two designs of trawl nets were towed simultaneously: a modified beam trawl with an 80 mm square mesh codend, and the industry standard beam trawl with an 80 mm diamond mesh codend. Twelve tows/area were conducted in July–August 2007. The catch was sorted into commercially important species (target and commercial unwanted catch) and discard species. Commercial organisms were counted, and discards were further sorted into benthic invertebrates and finfish and each were weighed.

(1) Wade O. Revill A.S. Grant A. & Sharp M. (2009) Reducing the discards of finfish and benthic invertebrates of UK beam trawlers. *Fisheries Research,* 97, 140–147.

6.42. Fit one or more soft, semi-rigid, or rigid grids or frames to trawl nets and use square mesh instead of a diamond mesh at the codend

- **One study** examined the effects of fitting one or more soft, semi-rigid, or rigid grids or frames to trawl nets and using a square mesh codend on subtidal benthic invertebrates. The study was in the Gulf of St Vincent[1] (Australia).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Unwanted catch abundance (1 study):** One replicated, paired, controlled study in Gulf of St Vincent[1] found that trawl nets fitted with a rigid U-shaped grid and a square-oriented mesh codend reduced the catch rates of three dominant groups of unwanted invertebrate catch species, compared to unmodified nets.

OTHER (1 STUDY)

- **Commercial catch abundance (1 study):** One replicated, paired, controlled study in the Gulf of St Vincent[1] found that trawl nets fitted with a rigid U-shaped grid and a square-oriented mesh codend reduced the catch rates of the commercially targeted western king prawn, due to reduced catch of less valuable smaller-sized prawns, compared to unmodified nets.

Background

Trawling is a method of fishing that involves pulling a cone-shaped fishing net (trawl) through the water behind one or more boats. The net is wide at the opening and narrows to a bag or 'codend', tied at the end with a drawstring, where organisms are trapped. Trawl nets can catch a considerable number of unwanted organisms, including non-commercially targeted species and organisms under the legal-size limit. Standard trawl nets are made from diamond-shaped mesh. To reduce the amount of unwanted organisms caught, one or more soft, semi-rigid or rigid grids or frames can be fitted to the inner side of the net before the codend, in combination with a codend made of square mesh instead of diamond mesh. The grid or frame is designed to prevent larger organisms, such as turtles, from entering the net/codend and allow smaller unwanted organisms to escape the codend, while retaining the commercially targeted organisms.

Evidence related to the use of grids in combination with other "bycatch reducing devices" are summarised under "Threat: Biological resource use – Fit one or more mesh escape panels/windows and one or more soft, rigid or semi-rigid grids or frames to trawl nets". Evidence related to the use of each modification (grids and square-mesh codend) separately, are summarised under "Threat: Biological resource use – Fit one or more soft, semi-rigid, or rigid grids or frames to trawl nets" and "Use a square mesh instead of a diamond mesh codend on trawl nets".

A replicated, paired, controlled study in 2012 in the Gulf of St Vincent, off the coast of South Australia (1) found that trawl nets fitted with a rigid U-shaped grid ("bycatch reduction device") and a square-oriented mesh codend resulted in lower catch rates of three dominant groups of unwanted invertebrate catch species, compared to unmodified nets. Compared to unmodified nets, the modified nets led to a 92% decrease in catch rate (kg/h) of sponges, 78–82% decrease in catch rate of crabs and other crustaceans, and a 61% decrease in catch rate of molluscs (excluding commercially valuable species of octopus, squid and cuttlefish; raw data not provided). A 15% decrease in catch rates of the commercially targeted western king prawn *Penaeus latisulcatus* was recorded due to reduced catch of less valuable smaller-sized prawns. In May 2012, unwanted catch of invertebrates in modified and unmodified nets were compared (see paper for details). Nets were deployed by four vessels during 29 paired hauls for 30 min (one modified; one unmodified; side-by-side simultaneously). All invertebrates were identified, sorted as commercial prawn catch or unwanted catch, and weighed.

(1) Gorman D. & Dixon C. (2015) Reducing discards in a temperate prawn trawl fishery: a collaborative approach to bycatch research in South Australia. *ICES Journal of Marine Science*, 72, 2609–2617.

6.43. Fit one or more mesh escape panels/windows to trawl nets and use a square mesh instead of a diamond mesh codend

- **One study** examined the effects of fitting one or more mesh escape panels to trawl nets and using a square mesh instead of a diamond mesh codend on subtidal benthic invertebrate populations. The study was in the English Channel[1] (UK).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Unwanted catch abundance (1 study):** One replicated, paired, controlled study in the English Channel[1] found that trawl nets fitted with two large square mesh release panels and a square mesh codend caught fewer unwanted catch of non-commercial invertebrates compared to standard trawl nets.

OTHER (1 STUDY)

- **Commercial catch abundance (1 study):** One replicated, paired, controlled study in the English Channel[1] found that trawl nets fitted with two large square mesh release panels and a square mesh codend caught fewer commercial shellfish, and fewer but more valuable commercially important fish, compared to standard trawl nets.

Background

Trawling is a method of fishing that involves pulling a cone-shaped fishing net (trawl) through the water behind one or more boats. The net is wide at the opening and narrows to a bag or 'codend', tied at the end with a drawstring, where organisms are trapped. Trawl nets can catch a considerable number of unwanted organisms, including non-commercially targeted species and organisms under the legal-size limit. Standard trawl nets are made from diamond-shaped mesh. To reduce the amount of unwanted organisms, a net can be modified by fitting one or more mesh "escape panels" in the outer mesh of the net before the codend, in combination with a codend made of square mesh instead of diamond mesh. These modifications are designed to allow smaller unwanted organisms to escape the main body of the net and the codend, while retaining the commercially targeted organisms.

Evidence related to the use of each modification (mesh panels and square-mesh codend) separately, are summarised under "Threat: Biological resource use – Fit one or more mesh escape panels/windows to trawl nets" and "Use a square mesh instead of a diamond mesh codend on trawl nets".

A replicated, paired, controlled study in 2007 in two areas of the English Channel, southwest England, UK (1) found that fishing with a trawl net fitted with a square mesh codend and two large square mesh release panels ("bycatch reduction devices") reduced the biomass of non-commercial unwanted invertebrate catch (discard) compared to a standard trawl. Across the two areas, the modified trawls caught 39–45% less invertebrate discard (349–730 kg) compared to the standard trawls (635–1,207 kg). However, they caught 22–82% fewer commercial shellfish (94–101 individuals), compared to standard trawls (120–570 individuals). The modified trawls also caught 22% less commercially important fish in one area, but those were worth more per kg than the fish caught in the standard trawls. Two designs of trawl nets were towed simultaneously: a modified beam trawl with an 80 mm square mesh codend and fitted with two large 200 mm square mesh release panels (one to release weed and one to release invertebrates), and the industry standard beam trawl with an 80 mm diamond mesh codend. Seven to nine tows/area were conducted in July–August 2007. The catch was sorted into commercially important species (target and non-target commercial catch) and discard species. Commercial organisms were counted, and discards were further sorted into benthic invertebrates and finfish and each were weighed.

(1) Wade O. Revill A.S. Grant A. & Sharp M. (2009) Reducing the discards of finfish and benthic invertebrates of UK beam trawlers. *Fisheries Research,* 97, 140–147.

6.44. Modify the design/attachments of a shrimp/prawn W-trawl net

- **One study** examined the effects of modifying the design/attachments of a W-trawl net used in shrimp/prawn fisheries on unwanted catch of subtidal benthic invertebrate. The study was in Moreton Bay[1] (Australia).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Unwanted catch overall abundance (1 study):** One replicated, paired, controlled study in Moreton Bay[1] found that four designs of W-trawl nets used in shrimp/prawn fisheries caught less non-commercial unwanted catch of crustaceans compared to a traditional Florida Flyer trawl net.

OTHERS (1 STUDY)

- **Commercial catch abundance (1 study):** One replicated, paired, controlled study in Moreton Bay[1] found that four designs of W-trawl nets used in shrimp/prawn fisheries caught lower amounts of the commercially targeted prawn species compared to a traditional Florida Flyer trawl net.

Background

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear coming into contact with the seabed (Collie *et al*. 2000). Trawling is a method of fishing that involves pulling a cone-shaped fishing net (trawl) through the water behind one or more boats. Trawling can be particularly damaging to benthic invertebrates as they are dragged along the seabed (mid-water trawls can also sometimes accidentally come into contact with the seabed). In addition, trawl nets, and in particular W-trawls used in the prawn/shrimp/nephrops fishery, can catch a considerable number of unwanted organisms, including non-commercially targeted species and organisms under the legal-size limit. To reduce the level of disturbance to the seabed and the amount of unwanted organisms caught, the design and/or attachments of the W-trawl net can be modified, for instance by securing the netting at the wing ends, pulling the top tongue of the net forward, modifying the attachment of the ground chain or by using a combination of such modifications (Balash *et al*. 2016).

Balash C., Sterling D. & Broadhurst M.K. (2016) Progressively evaluating a penaeid W trawl to improve eco-efficiency. *Fisheries Research*, 181, 148–154.

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

A replicated, paired, controlled study in 2014 in Moreton Bay, Australia (1) found that four designs of W-trawl nets caught less non-commercial unwanted catch of crustaceans (discard) compared to a traditional Florida Flyer trawl net. All designs of W-trawls caught smaller amounts of crustacean discard than the traditional trawl (design 1: 1.5 vs Florida Flyer: 5.2 kg/ha; design 2: 5.6 vs 7.6; design 3: 4.9 vs 6.7; design 4: 6.9 vs 9.4). All designs of W-trawl caught lower amounts of the commercially targeted prawn species compared to the traditional trawl (27–39% reductions). In February 2014, unwanted catch from four W-trawl designs were compared to that of the Florida Flyer trawl during paired simultaneous 15–60 min deployments (one net of either one of the four designs on one side of the vessel, one Florida Flyer net on the other; 10–13 deployments/design). Design 1: unmodified W-trawl. Design 2: W-trawl with secured netting at the wing ends. Design 3: design 2 with the top tongue pulled forward and one chain link removed from each side of the ground chain. Design 4: design 3 further modified at wing ends (fitting "Dan lenos"). See paper for technical details. All nets were fitted with batwing otter boards, a "turtle-excluder device" (escape panel), and a "bycatch reducing device" ("fisheye"). At the end of each haul, catches were sorted into commercially targeted catch, commercial unwanted catch (large crabs, squid and octopus), and crustacean discard, and all were weighed.

(1) Balash C., Sterling D. & Broadhurst M.K. (2016) Progressively evaluating a penaeid W trawl to improve eco-efficiency. *Fisheries Research,* 181, 148–154.

6.45. Reduce the number or modify the arrangement of tickler chains/chain mats on trawl nets

- **Three studies** examined the effects of reducing the number or modifying the arrangement of tickler chains/chain mats on subtidal benthic invertebrates. All were in the North Sea[1,2a–b] (Germany and Netherlands).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (3 STUDIES)

- **Overall abundance (1 study):** One replicated, paired, controlled study in the North Sea[1] found that using a beam trawl with a chain mat caused lower mortality of benthic invertebrates in the trawl tracks compared to using a beam trawl with tickler chains.
- **Unwanted catch abundance (2 studies):** One of two replicated, paired, controlled studies in the North Sea[2a,b] found that all three modified parallel tickler chain arrangements reduced the combined amount of non-commercial unwanted invertebrate and fish catch compared to unmodified trawl nets[2b], but the other found that none of three modified parabolic tickler chain arrangements reduced it[2a].

OTHER (2 STUDIES)

- **Commercial catch abundance (2 studies):** One of two replicated, paired, controlled studies in the North Sea[2a,b] found that three modified parabolic tickler chain arrangements caught similar amounts of commercial species to unmodified nets[2a], but the other[2b] found that three modified parallel tickler chain arrangements caught lower amounts.

Background

Trawling is a method of fishing that involves pulling a fishing net (trawl) through the water behind one or more boats. Beam trawls are rigged with either tickler chains stretching from one side of the trawl mouth to the other, or a stone mat. Tickler chains are metal chains which drag along the seabed in front of the net with the aim of disturbing fish in the path of the trawl, causing them to enter the net. A chain mat is a network of chains over the trawl's mouth designed to prevent large stones from entering the gear while also disturbing fish that are then caught in the net. Chain and mats can therefore disturb the seabed as well as increase unwanted catch of subtidal benthic invertebrates. To potentially reduce these unwanted effects, fewer tickler chains can be used, or the arrangement of chains and mats can be modified (Van Marlen *et al.* 2005), for example by attaching only one end of the chain to the beam, or reducing the size of the chain links (Bergman & Van Santbrink 2000; Broadhurst *et al.* 2015).

Evidence for other interventions related to fishing gear modifications is summarised under "Threat: Biological resource use".

Bergman M.J.N. & Van Santbrink J.W. (2000) Mortality in megafaunal benthic populations caused by trawl fisheries on the Dutch continental shelf in the North Sea in 1994. *ICES Journal of Marine Science* 57, 1321–1331.

Broadhurst M.K., Sterling D.J. & Millar R.B. (2015) Traditional vs. novel ground gears: Maximising the environmental performance of penaeid trawls. *Fisheries Research,* 167, 199–206.

Van Marlen B., Bergman M.J.N., Groenewold S. & Fonds M. (2005) New approaches to the reduction of non-target mortality in beam trawling. *Fisheries Research,* 72, 333–345.

A replicated, paired, controlled study in 1992–1995 in one area of sandy seabed in the south-eastern North Sea, Netherlands and Germany (1) found that using a beam trawl with a chain mat caused lower mortality of benthic invertebrates in the trawl tracks (not caught by the nets) compared to using a beam trawl with tickler chains. Mortality using a chain mat varied between 4 and 15% depending on species and was lower than when using tickler chains (1–30%). In spring-summer 1992–1995 parallel strips (2,000 x 60 m, 300 m apart, number unspecified) were fished with 4-m beam trawls with either a chain mat or tickler chains. Prior to trawling, 'mega'-invertebrates >10 mm) and 'macro'-invertebrates (between 1 and 10 mm) were counted from samples taken from each strip using a dredge and a sediment grab. After 24–48 h following trawling, all strips were sampled again using the same methods. Mortality (from trawling) of invertebrates present in the trawl tracks was calculated using the difference between the before and after-trawling abundances (assuming all animals killed by trawling had been eaten by predators).

A replicated, paired, controlled study in 1999 in one area of seabed in the North Sea, Netherlands (2a) found that none of three modified tickler chain arrangements for trawl nets reduced the amount of non-commercial unwanted invertebrates and fish catch (discard), compared to unmodified trawl nets. Nets modified with two of the three tickler chain arrangements tested caught similar amount of discard to unmodified nets (153–175 vs 145–166 kg/h). The third arrangement (25 cm spacing) caught more discard than unmodified nets (123 vs 112 kg/h). All modified nets caught similar amounts of commercial species to unmodified nets (35–50 vs 33–36 kg/h). In conventional tickler chain rigging, both ends of chains are attached at either ends of the beam. Three parabolic tickler chain arrangements, where attachment points are distributed along the beam, were tested on trawl nets: 25 cm spacing; 40 cm spacing; 25 cm spacing with 35 cm for the centre chain. In October 1999, each arrangement was compared a conventional tickler chain during 5–17 paired simultaneous deployments along parallel strips (2,000 x 30 m). Catches were sorted into commercial and discard species, and each group weighed.

A replicated, paired, controlled study in 1999 in one area of seabed in the North Sea, Netherlands (2b) found that all three modified tickler chain arrangements for trawl nets reduced the amount of non-commercial unwanted invertebrates and fish catch (discard), compared to unmodified trawl nets. Nets modified with either of three tickler chain arrangements tested caught less discard than unmodified nets (46–80 vs 80–117 kg/h). However, all modified nets also caught lower amounts of commercial species compared to unmodified nets (43–49 vs 52–58 kg/h). In conventional tickler chain rigging, both ends of chains are attached at either ends of the beam. Three parallel tickler chain arrangements, where chains are distributed along the beam but only attached at one end, were tested on trawl nets: 21 chains, 50 cm spacing; 29 chains, 35 cm spacing; 29 chains, 35 cm spacing with 10 connected pairs. In March–April 1999, each arrangement was compared to a conventional tickler chain during 11–42 paired simultaneous deployments along parallel strips (2,000 x 30 m). Catches were sorted into commercial and discard species, and each group weighed.

(1) Bergman M.J.N. & Van Santbrink J.W. (2000) Mortality in megafaunal benthic populations caused by trawl fisheries on the Dutch continental shelf in the North Sea in 1994. *ICES Journal of Marine Science,* 57, 1321–1331.
(2a-b) Van Marlen B., Bergman M.J.N., Groenewold S. & Fonds M. (2005) New approaches to the reduction of non-target mortality in beam trawling. *Fisheries Research,* 72, 333–345.

6.46. Use a larger mesh size on trammel nets

- **One study** examined the effects of using a larger mesh size on trammel nets on subtidal benthic invertebrates. The study was in the North Atlantic Ocean[1] (Portugal).

COMMUNITY RESPONSE (1 STUDY)

- **Unwanted catch community composition (1 study):** One replicated, controlled, study in the North Atlantic Ocean[1] found that using larger mesh sizes in the inner and/or outer panels of trammel nets did not affect the community composition of unwanted catch of non-commercial invertebrates (discard).

POPULATION RESPONSE (1 STUDY)

- **Unwanted catch abundance (1 study):** One replicated, controlled, study in the North Atlantic Ocean[1] found that using larger mesh sizes in the inner and/or outer panels of trammel nets did not reduce the abundance of unwanted catch of non-commercial invertebrates (discard).

Background

Some fisheries use static nets, such as trammel nets, that are left in the water to passively catch the commercially targeted species. Trammel nets are a wall of netting, usually comprised of three layers: a slack central layer with a small mesh sandwiched between two outer layers with a much larger mesh. The net is kept vertical by floats on the headrope and weights on the bottom rope. Trammels are considered to have less environmental impact than trawl nets, as they have minimal contact with the seabed and are more species and size-selective, however they have been shown to also result in some unwanted catch of invertebrates (Erzini *et al.* 2006; Gonçalves *et al.* 2007). To reduce the amount of unwanted catch in trammel net fishing, the mesh size can be increased, to potentially allow more organisms to escape (Gonçalves *et al.* 2008).

Erzini K., Gonçalves J.M., Bentes L., Moutopoulos D.K., Casal J.A.H., Soriguer M.C., Puente E., Errazkin L.A. & Stergiou K.I. (2006) Size selectivity of trammel nets in southern European small-scale fisheries. *Fisheries Research*, 79, 183–201.

Gonçalves J.M.S., Bentes L., Coelho R., Monteiro P., Ribeiro J., Correia C., Lino P.G. & Erzini K. (2008) Non-commercial invertebrate discards in an experimental trammel net fishery. *Fisheries Management and Ecology*, 15, 199–1210.

Gonçalves J.M.S., Stergiou K.I., Hernando J.A., Puente E., Moutopoulos D.K., Arregi L., Soriguer M.C., Vilas C., Coelho R. & Erzini K. (2007) Discards from experimental trammel nets in southern European small-scale fisheries. *Fisheries Research*, 88, 5–14.

A replicated, controlled study in 2001 off the coast of Algarve, southern Portugal, North Atlantic Ocean (1) found that using larger mesh sizes in the inner and/or outer panels of trammel nets did not affect the community composition or reduce the abundance of unwanted catch of non-commercial invertebrates (discard). Discard community composition was similar in all six mesh-size configurations tested (data presented as statistical model results and graphical analyses). This was also true for their abundance which ranged from 21 to 29 individuals/1,000 m of net (corresponding to 39–54% of the total catch). Between January and December 2000, six trammel net configurations were tested during 40 fishing trials. Each configuration consisted of a combination of one of two sizes of large-mesh outer panels (600 or 800 mm) and one of three small-mesh inner panels (100, 120, or 140 mm). A total of 150 nets were deployed in groups (30 nets/group). For each group, five nets of each configuration were joined by a footrope (2 m gap between each net). For each configuration, catches were sorted into commercial (fishery target species and commercial bycatch) and unwanted non-commercial species (invertebrate discards), identified and counted. Commercial catch data for each configuration were not reported.

(1) Gonçalves J.M.S., Bentes L., Coelho R., Monteiro P., Ribeiro J., Correia C., Lino P.G. & Erzini K. (2008) Non-commercial invertebrate discards in an experimental trammel net fishery. *Fisheries Management and Ecology*, 15, 199–1210.

6.47. Use traps instead of fishing nets

- **One study** examined the effects of using traps instead of fishing nets on subtidal benthic invertebrates. The study took place in the Mediterranean Sea[1] (Spain).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Unwanted catch abundance (1 study):** One replicated, controlled study in the Mediterranean Sea[1] found that the combined amount of unwanted catch of invertebrates and fish appeared lower using plastic traps than trammel nets, but higher using collapsible traps.

OTHER (1 STUDY)

- **Commercial catch abundance (1 study):** One replicated, controlled study in the Mediterranean Sea[1] found that the catch of commercially targeted lobsters was lower using traps than in trammel nets.

Background

Fishing nets can have poor selectivity, leading to high amounts of unwanted catch, and also can negatively impact the seabed and benthic communities due to physical contact and disturbance (Amengual-Ramis *et al*. 2016). Although fishing nets and traps usually target different species and are used in different fisheries, in certain cases, they can target and catch the same species. Replacing fishing nets with traps and pots where feasible can help reduce the impacts on invertebrate populations, through a reduction in physical disturbances and a reduction in unwanted catch. For instance, trammel nets used in lobster fisheries could be replaced with traps (Amengual-Ramis *et al*. 2016).

Evidence for other interventions related to trap and pot fishery is summarised under "Threat: Biological resource use – Modify the position of traps", "Modify the design of traps", "Use a different bait species in traps", "Fit one or more soft, semi-rigid, or rigid grids or frames on pots and traps" and "Fit one or more mesh escape panels/windows on pots and traps".

Amengual-Ramis J.F., Vazquez-Archdale M., Canovas-Perez C. & Morales-Nin B. (2016) The artisanal fishery of the spiny lobster *Palinurus elephas* in Cabrera National Park, Spain: comparative study on traditional and modern traps with trammel nets. *Fisheries Research*, 179, 23–32.

A replicated, controlled study in 2011–2012 of seabed composed of mud, kelp, and maërl, off the southeastern coast of Mallorca, Mediterranean Sea, Spain (1) found that experimental designs of lobster traps appeared to catch different combined amount of non-commercial unwanted invertebrates and fish (discard) than commercially used trammel nets, but the amount varied with trap design. Data were not statistically tested. When comparing similar length-deployment for each fishing design, the amount of discard caught in plastic traps (3 individuals/450 m) tended to be lower than in trammel nets (5.7), but higher in collapsible traps (16). Catches of legal-size commercially targeted lobsters tended to be lower in traps (0–0.3 lobsters/450 m) than in trammel nets (1.3). In May–September 2011, traps (900/design) were deployed at 50–100 m depth for 24h (see paper for details of each design). Lobsters and unwanted species caught were counted and measured in each trap. Baited traps were deployed in two 450 m-long strings of 30 traps each (one line/design; >200 m apart). In May–August 2012, similar data for trammel nets were obtained onboard commercial vessels (119 nets, 50 m each, deployed overnight).

(1) Amengual-Ramis J.F., Vazquez-Archdale M., Canovas-Perez C. & Morales-Nin B. (2016) The artisanal fishery of the spiny lobster *Palinurus elephas* in Cabrera National Park, Spain: comparative study on traditional and modern traps with trammel nets. *Fisheries Research*, 179, 23–32.

6.48. Modify the design of traps

- **Two studies** examined the effects of modifying the design of traps on subtidal benthic invertebrates. One study took place in the Mediterranean Sea[1] (Spain), and one in the South Pacific Ocean[2] (New Zealand).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Unwanted catch abundance (2 studies):** Two replicated, controlled studies in the Mediterranean Sea[1] and the South Pacific Ocean[2] found that the amount of combined unwanted catch of invertebrates and fish varied with the type of trap design used[1,2] and the area[2].

OTHER (1 STUDY)

- **Commercial catch abundance (1 study):** One replicated, controlled study in the Mediterranean Sea[1] found that plastic traps caught some legal-size commercially targeted lobsters while collapsible traps caught none.

Background

Traps or pots are static gears often used to fish for crabs or lobsters. They consist of structures into which species of commercial interest enter through funnels. These funnels encourage entry but limit escape, and often catch a large amount of unwanted species (Stevens 1996). Trap design can be modified, such as by using different shape or material for the frame (Amengual-Ramis *et al.* 2016; Major *et al.* 2017), to potentially reduce the amount of subtidal benthic invertebrate bycatch (Arrasate-López *et al*. 2012; Schoeman *et al*. 2002).

Evidence for related interventions is summarised under "Threat: Biological resource use – Modify the position of traps" and "Use different bait species in traps".

Amengual-Ramis J.F., Vazquez-Archdale M., Canovas-Perez C. & Morales-Nin B. (2016) The artisanal fishery of the spiny lobster *Palinurus elephas* in Cabrera National Park, Spain: comparative study on traditional and modern traps with trammel nets. *Fisheries Research*, 179, 23–32.

Arrasate-López M., Tuset V.M., Santana J.I., García-Mederos A., Ayza O. & González J.A. (2012) Fishing methods for sustainable shrimp fisheries in the Canary Islands (North-West Africa). *African Journal of Marine Science*, 34, 331–339.Major R.N., Taylor D.I., Connor S., Connor G. & Jeffs A.G. (2017) Factors affecting bycatch in a developing New Zealand scampi potting fishery. *Fisheries Research*, 186, 55–64.

Stevens B.G. (1996) Crab bycatch in pot fisheries. *Solving bycatch: considerations for today and tomorrow*, 151–158.

Schoeman D.S., Cockcroft A.C., Van Zyl D.L. & Goosen P.C. (2002) Changes to regulations and the gear used in the South African commercial fishery for *Jasus lalandii. South African Journal of Marine Science*, 24, 365–369.

A replicated, controlled study in 2011–2012 of seabed composed of mud, kelp, and maërl, off the southeastern coast of Mallorca, western Mediterranean Sea, Spain (1) found that plastic lobster traps appeared to catch lower amounts of non-commercial unwanted catch (discard) than collapsible traps. Data were not statistically tested. The amount of discard caught in plastic traps (3 individuals/450 m) tended to be lower than in collapsible traps (16). In addition, plastic traps caught some legal-size commercially targeted lobsters (0.3/450 m), while collapsible traps caught none. In May–September 2011, two new designs of traps, plastic and collapsible (900/design), were deployed at 50–100 m depth for 24 h (see original paper for details of each design). Lobsters and discard species caught were counted and measured in each trap. Baited traps were deployed in two 450 m-long strings of 30 traps each (one line/design; >200 m apart).

A replicated, controlled study in 2014–2015 in two areas of seabed in the South Pacific Ocean, New Zealand (2) found that four different trap designs used to catch New Zealand scampi *Metanephros challengeri* caught different amount of unwanted catch of combined invertebrates and fish, but the effects varied between areas. In one area, rectangular traps caught more unwanted catch (2 individuals/trap) than box traps and standard traps (1 individual/trap; no difference between the two designs). In the other site, rectangular traps caught more unwanted catch (8 individuals/trap) than boxed traps (3 individuals/trap), and both caught more than domed plastic traps and standard traps (1 individual/trap; no difference between the two designs). Four different trap designs were tested in two areas: a standard creel trap, a box shaped creel trap, a rectangular shaped creel trap and a domed plastic trap. At Chatham Rise from November–December 2014, three designs (rectangular, box, standard) were tested during three deployments (three 500 m lines of 30 baited traps/deployment; 10 traps/design/line). At Cape Palliser in April 2015, all four designs were tested during three deployments (one 500 m line of 30 baited traps/deployment; 7–10 traps/design/line). Traps were recovered after 18 hours, and unwanted catch identified and counted.

(1) Amengual-Ramis J.F., Vazquez-Archdale M., Canovas-Perez C. & Morales-Nin B. (2016) The artisanal fishery of the spiny lobster *Palinurus elephas* in Cabrera National Park, Spain: comparative study on traditional and modern traps with trammel nets. *Fisheries Research*, 179, 23–32.
(2) Major R.N., Taylor D.I., Connor S., Connor G. & Jeffs A.G. (2017) Factors affecting bycatch in a developing New Zealand scampi potting fishery. *Fisheries Research*, 186, 55–64.

6.49. Modify the position of traps

- **Two studies** examined the effects of modifying the position of traps on subtidal benthic invertebrate populations. One study was in the Varangerfjord[1] (Norway), the other in the North Atlantic Ocean[2] (Spain).

COMMUNITY RESPONSE (1 STUDY)

- **Unwanted catch species richness/diversity (1 study):** One replicated, controlled study in the North Atlantic[2] found that semi-floating traps caught fewer unwanted catch species compared to standard bottom traps.

POPULATION RESPONSE (2 STUDIES)

- **Unwanted catch abundance (2 studies):** Two replicated, controlled studies in the Varangerfjord[1] and the North Atlantic[2] found that floating or semi-floating traps caught fewer unwanted invertebrates compared to standard bottom traps.

OTHER (2 STUDIES)

- **Commercial catch abundance (2 studies):** Two replicated, controlled studies in the Varangerfjord[1] and the North Atlantic[2] found that floating or semi-floating traps caught similar amounts (abundance and biomass) of commercially targeted species as standard bottom traps.

Background

Traps or pots are static gears often used to fish for crabs or lobsters. They consist of structures into which species of commercial interest enter through funnels. These funnels encourage entry but limit escape, and often catch a large amount of unwanted catch species (Stevens 1996). The position of traps can be modified, for example by floating traps in the water above the seabed rather than placing them static on the seabed (Furevik *et al.* 2008), to potentially reduce the amount of certain species of subtidal benthic invertebrate accidentally caught.

Evidence for related interventions is summarised under "Threat: Biological resource use – Modify the design of traps" and "Use different bait species in traps".

Furevik D.M., Humborstad O.B., Jørgensen T. & Løkkeborg S. (2008) Floated fish pot eliminates bycatch of red king crab and maintains target catch of cod. *Fisheries Research*, 92, 23–27.

Stevens B.G. (1996) Crab bycatch in pot fisheries. *Solving bycatch: considerations for today and tomorrow*, 151–158.

A replicated, controlled study in 2003–2004 in the Varangerfjord, Norway (1) found that traps floated above the seabed caught fewer unwanted red king crabs *Paralithodes camtschaticus*, compared to standard groundfish traps. Red king crabs were only found in two of the 73 floated traps (2 and 3 crabs/trap), while all 77 standard traps caught crabs with an average catch of 21 crabs/trap. There was no difference in the number of marketable catches of the commercially targeted species, cod *Gadus morhua*, between the two trap designs. In August–September 2003 and 2004, sixteen lines of baited traps (100 x 150 x 120 cm) were deployed at 70–250 m depths. Two types of trap were used: a standard two-chamber groundfish trap and a floated version (approximately 70 cm above the seabed) of the same trap. Each line held five traps/design, placed alternatively. The traps were recovered after 24 hours, and catches sorted and counted. In this study, floating traps were used to reduce clogging of the traps by unwanted red king crabs and improve catch efficiency of cod, rather than to conserve red king crab.

A replicated, controlled study in 2003–2004 at four different water depths in areas of rocky seabed around the Canary Islands, North Atlantic Ocean, Spain (2) found that using semi-floating shrimp traps instead of traditional bottom traps appeared to reduce the catch and biomass of unwanted non-commercial species (discards) and unwanted commercial species (here referred to as bycatch), consistently across water depths. Results were not tested for statistical significance. Across water depths, semi-floating traps tended to catch fewer discard species of lower biomass (1–3 species; 0.006–0.6 g/trap/day) compared to bottom traps (2–4 species; 1–23 g/trap/day), and fewer bycatch species of lower biomass (semi-floating: 0–4 species, 0–18 g/trap/day; bottom: 1–6 species, 59–363 g/trap/day). The overall number and biomass of commercially targeted prawn species caught tended to be similar using semi-floating traps (2–6 species; 20.5–135 g/trap/day) and bottom traps (3–5 species; 16.6–107.3 g/trap/day), but the trap types caught different species. Four surveys were undertaken between October 2003 and October 2004. During each survey, an unspecified number of baited bottom traps and semi-floating traps (2 m above the seabed) were deployed at 100 m depth intervals between 120 and 1,300 m depths for 15–25 h. The number and biomass of bycatch, discard, and commercially targeted species were recorded. Data for a total of 487 bottom traps and 1,971 semi-floating traps were collected.

(1) Furevik D.M., Humborstad O.B., Jørgensen T. & Løkkeborg S. (2008) Floated fish pot eliminates bycatch of red king crab and maintains target catch of cod. *Fisheries Research,* 92, 23–27.
(2) Arrasate-López M., Tuset V.M., Santana J.I., García-Mederos A., Ayza O. & González J.A. (2012) Fishing methods for sustainable shrimp fisheries in the Canary Islands (North-West Africa). *African Journal of Marine Science*, 34, 331–339.

6.50. Use different bait species in traps

- **One study** examined the effects of using different bait species in traps on subtidal benthic invertebrates. The study took place in the South Pacific Ocean[1] (New Zealand).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Unwanted catch abundance (1 study):** One replicated, controlled study in the South Pacific Ocean[1] found that the type of bait used in fishing pots did not change the quantity of unwanted invertebrates caught.

Background

Traps or pots are static gears often used to fish for crabs or lobsters. They consist of structures into which species of commercial interest enter through funnels. These funnels encourage entry but limit escape, and often catch a large amount of unwanted catch species (Stevens 1996). Traps or pots can also be baited to further encourage entry. By using a different bait species, for instance one that is less attractive to certain unwanted catch species, the amount of unwanted catch can potentially be reduced (Major *et al*. 2017). Evidence for other interventions related to reducing accidental and/or unwanted catch in trap and pot fishery is summarised under "Threat: Biological resource use – Modify the position of traps", "Modify the design of traps", "Fit one or more soft, semi-rigid, or rigid grids or frames on pots and traps" and "Fit one or more mesh escape panels/windows on pots and traps".

Major R.N., Taylor D.I., Connor S., Connor G. & Jeffs A.G. (2017) Factors affecting bycatch in a developing New Zealand scampi potting fishery. *Fisheries Research*, 186, 55–64.
Stevens B.G. (1996) Crab bycatch in pot fisheries. *Solving bycatch: considerations for today and tomorrow*, 151–158.

A replicated, controlled study in 2014–2015 in two seabed areas in the South Pacific Ocean, New Zealand (1) found that the type of bait used in the New Zealand scampi *Metanephros challengeri* pot fishery did not change the quantity of unwanted invertebrates caught, in either area. The quantity of unwanted invertebrates caught was similar in pots baited with mackerel *Scomber australasicus*, barracouta *Thyrsites atun*, or squid *Nototodarus sloanii* (abundance data not shown). In two areas, three bait species were tested: mackerel vs squid, and barracouta vs squid (mackerel vs barracouta not tested). At Chatham Rise from November–December 2014, traps baited with either mackerel or squid (equal number of traps) were tested during three deployments (three 500 m lines of 30 traps/deployment). At Cape Palliser in April 2015, traps baited with either barracouta or squid (equal number of traps) were tested during three deployments (one 500 m line of 30 traps/deployment). Traps were recovered after 18 hours, and unwanted invertebrate catch identified and counted.

(1) Major R.N., Taylor D.I., Connor S., Connor G. & Jeffs A.G. (2017) Factors affecting bycatch in a developing New Zealand scampi potting fishery. *Fisheries Research*, 186, 55–64.

6.51. Fit one or more soft, semi-rigid, or rigid grids or frames on pots and traps

- **One study** examined the effects of fitting one or more soft, semi-rigid, or rigid grids or frames on pots and traps on subtidal benthic invertebrates. The study took place in the Corindi River system[1] (Australia).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Unwanted catch abundance (1 study):** One replicated, controlled study in the Corindi River system[1] found that traps fitted with escape frames appeared to reduce the proportion of unwanted undersized mud crabs caught, compared to conventional traps without escape frames.

Background

Traps or pots are static gears often used to fish for crabs or lobsters. They consist of structures into which species of commercial interest enter through funnels. These funnels encourage entry but limit escape, and often catch a large amount of unwanted species (Stevens 1996). To try to minimise the amount of unwanted catch from this type of fishing, g a device such as a rigid frame or rigid wires can be fitted to the entrance of the trap, or other types of "excluder devices" can be used, to reduce the likelihood of unwanted species entering, but also to allow small unwanted species to escape (Broadhurst *et al.* 2014). Such devices have been effective in reducing accidental catches of seabirds (Morris *et al*. 2011), seals (Königson *et al.* 2015), and terrapins (Roosenburg & Green 2000), and therefore may be considered when trying to reduce unwanted catch of subtidal benthic invertebrate species.

Evidence related to the use of other "bycatch reduction devices"/"excluder devices" on pots and traps are summarised under "Threat: Biological resource use – Fit one or more mesh escape panels/windows on pots and traps".

Broadhurst M.K., Butcher P.A. & Cullis B.R. (2014) Effects of mesh size and escape gaps on discarding in an Australian giant mud crab (*Scylla serrata*) trap fishery. *PloS One*, 9, e106414.
Königson S., Lövgren J., Hjelm J., Ovegård M., Ljunghager F. & Lunneryd S. G. (2015) Seal exclusion devices in cod pots prevent seal bycatch and affect their catchability of cod. *Fisheries Research*, 167, 114–122.
Morris A.S., Wilson S.M., Dever E.F. & Chambers R.M. (2011) A test of bycatch reduction devices on commercial crab pots in a tidal marsh creek in Virginia. *Estuaries and Coasts*, 34, 386–390.
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.
Stevens B.G. (1996) Crab bycatch in pot fisheries. *Solving bycatch: considerations for today and tomorrow*, 151–158.

A replicated, controlled study (date unspecified but appears to be 2012) in a muddy and sandy area in the Corindi River system, eastern Australia (1) found that traps used to catch giant mud crabs *Scylla serrata* appeared to catch fewer undersized mud crabs when fitted with escape frame, compared to conventional traps without escape frames. The proportion of undersized crabs caught in traps fitted with frames appeared lower (2%) compared to conventional traps (29%; results not tested for statistical significance). In addition, the number of wounded mud crabs (undersized and commercial size) was statistically similar in traps with escape frames (0.06 crabs/trap) and conventional traps (0.13 crabs/trap). Conventional traps have four 300 × 200 mm funnel entrances but no escape frames. Conventional traps were modified by fitting two 46 × 120 mm escape frames. Seven modified traps and seven conventional traps were tested during 20 deployments. All traps were baited with sea mullet *Mugil cephalus*. Traps were recovered after 24 hours, and all catch identified, counted, and any wounds assessed.

(1) Broadhurst M.K., Butcher P.A. & Cullis B.R. (2014) Effects of mesh size and escape gaps on discarding in an Australian giant mud crab (*Scylla serrata*) trap fishery. *PloS One*, 9, e106414.

6.52. Fit one or more mesh escape panels/windows on pots and traps

- We found no studies that evaluated the effects of fitting one or more mesh escape panels/windows on pots and traps on subtidal benthic invertebrate 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

Traps or pots are static gears often used to fish for crabs or lobsters. They consist of structures into which species of commercial interest enter through funnels. These funnels encourage entry but limit escape, and often catch a large amount of unwanted species (Stevens 1996). To try to minimise the amount of unwanted catch from this type of fishing, a device such as an escape zone can be fitted at the back or sides of the trap, or other types of "bycatch reducing devices" used, to increase the likelihood of unwanted species escaping. Such devices may potentially help reduce unwanted catch of subtidal benthic invertebrate species and benefit their populations. Evidence related to the use of other "bycatch reduction devices" on pots and traps are summarised under "Threat: Biological resource use – Fit one or more soft, semi-rigid, or rigid grids or frames on pots and traps".

Stevens B.G. (1996) Crab bycatch in pot fisheries. *Solving bycatch: considerations for today and tomorrow*, 151–158.

6.53. Increase the mesh size of pots and traps

- **One study** examined the effects of increasing the mesh size of pots and traps on subtidal benthic invertebrates. The study took place in the Corindi River system[1] (Australia).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Unwanted catch abundance (1 study):** One replicated, controlled study in the Corindi River system[1] found that traps designed with larger mesh appeared to reduce the proportion of unwanted undersized mud crabs caught, compared to conventional traps of smaller mesh.

Background

Traps or pots are static gears often used to fish for crabs or lobsters. They consist of structures into which species of commercial interest enter through funnels. These funnels encourage entry but limit escape, and often catch a large amount of unwanted species (Stevens 1996). To try to minimise the amount of unwanted catch from this type of fishing, the size of the mesh used to construct the pots/traps can be increased, to increase the likelihood of unwanted species or smaller/younger individuals of the economically targeted species escaping (Broadhurst *et al.* 2014). This may potentially help reduce unwanted catch of subtidal benthic invertebrate species and benefit their populations.

Evidence related to the use of other "bycatch reduction devices" on pots and traps are summarised under "Threat: Biological resource use – Fit one or more soft, semi-rigid, or rigid grids or frames on pots and traps".

Broadhurst M.K., Butcher P.A. & Cullis B.R. (2014) Effects of mesh size and escape gaps on discarding in an Australian giant mud crab (*Scylla serrata*) trap fishery. *PloS One*, 9, e106414.
Stevens B.G. (1996) Crab bycatch in pot fisheries. *Solving bycatch: considerations for today and tomorrow*, 151–158

A replicated, controlled study (date unspecified but appears to be 2012) in a muddy and sandy area in the Corindi River system, eastern Australia (1) found that traps used to catch giant mud crabs *Scylla serrata* appeared to catch fewer unwanted undersized mud crabs when designed with larger mesh size, compared to conventional traps. The proportion of undersized crabs caught in traps with 101 mm mesh appeared lower (22%) compared to conventional traps with 51 mm mesh (29%; results not tested for statistical significance). In addition, the number of wounded mud crabs (undersized and commercial size) was statistically lower in traps with larger mesh size (0.03 crabs/trap) compared to conventional traps (0.13 crabs/trap). Conventional traps are designed with 51 mm mesh. Conventional traps were modified by increasing the mesh size to 101 mm. Seven modified traps and seven conventional traps were tested during 20 deployments. All traps were baited with sea mullet *Mugil cephalus*. Traps were recovered after 24 hours, and all catch identified, counted, and any wounds assessed.

(1) Broadhurst M.K., Butcher P.A. & Cullis B.R. (2014) Effects of mesh size and escape gaps on discarding in an Australian giant mud crab (*Scylla serrata*) trap fishery. *PloS One*, 9, e106414.

6.54. Fit one or more soft, semi-rigid, or rigid grids or frames and increase the mesh size of pots and traps

- **One study** examined the effects of fitting one or more soft, semi-rigid, or rigid grids or frames and increasing the mesh size of pots and traps on subtidal benthic invertebrates. The study took place in the Corindi River system[1] (Australia).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Unwanted catch abundance (1 study):** One replicated, controlled study in the Corindi River system[1] found that traps fitted with escape frames and designed with larger mesh appeared to reduce the proportion of unwanted undersized mud crabs caught, compared to conventional traps without escape frames and smaller mesh.

Background

Traps or pots are static gears often used to fish for crabs or lobsters. They consist of structures into which species of commercial interest enter through funnels. These funnels encourage entry but limit escape, and often catch a large amount of unwanted species (Stevens 1996). To try to minimise the amount of unwanted catch from this type of fishing, a device such as a rigid frame or rigid wires can be fitted to the entrance of the trap or elsewhere on the trap, to reduce the likelihood of large unwanted species entering, but also to allow small unwanted species to escape once inside the traps (Broadhurst *et al.* 2014). In combination with this frame, the size of the mesh used to construct the pots/traps can be increased, to increase the likelihood of unwanted species or smaller/younger individuals of the economically targeted species escaping (Broadhurst *et al.* 2014). This may potentially help reduce unwanted catch of subtidal benthic invertebrate species and benefit their populations.

Evidence related to the use of other "bycatch reduction devices" on pots and traps are summarised under "Threat: Biological resource use – Fit one or more soft, semi-rigid, or rigid grids or frames on pots and traps" and "Increase the mesh size of pots and traps".

Broadhurst M.K., Butcher P.A. & Cullis B.R. (2014) Effects of mesh size and escape gaps on discarding in an Australian giant mud crab (*Scylla serrata*) trap fishery. *PloS One*, 9, e106414.

Stevens B.G. (1996) Crab bycatch in pot fisheries. *Solving bycatch: considerations for today and tomorrow*, 151–158

A replicated, controlled study (date unspecified but appears to be 2012) in a muddy and sandy area in the Corindi River system, eastern Australia (1) found that traps used to catch giant mud crabs *Scylla serrata* appeared to catch fewer unwanted undersized mud crabs when fitted with escape frames and designed with larger mesh size, compared to conventional traps. The proportion of undersized crabs caught in traps fitted with frames and designed with 101 mm mesh appeared lower (11%) compared to conventional traps without frames and of 51 mm mesh (29%; results not tested for statistical significance). In addition, the number of wounded mud crabs (undersized and commercial size) was statistically similar in traps with escape frames and larger mesh size (0.04 crabs/trap) and conventional traps (0.13 crabs/trap). Conventional traps have four 300 × 200 mm funnel entrances, no escape frames, and are designed with 51 mm mesh. Conventional traps were modified by fitting two 46 × 120 mm escape frames and increasing the mesh size to 101 mm. Seven modified traps and seven conventional traps were tested during 20 deployments. All traps were baited with sea mullet *Mugil cephalus*. Traps were recovered after 24 hours, and all catch identified, counted, and any wounds assessed.

(1) Broadhurst M.K., Butcher P.A. & Cullis B.R. (2014) Effects of mesh size and escape gaps on discarding in an Australian giant mud crab (*Scylla serrata*) trap fishery. *PloS One*, 9, e106414.

6.55. Release live unwanted catch first before handling commercial species

- We found no studies that evaluated the effects of releasing live unwanted catch first before handling commercial species on subtidal benthic invertebrate 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

Trawling can lead to the capture large amounts of unwanted catch species. Animals caught in trawling nets can die from injuries sustained in the net, during handling, or when the catch is processed (Revill & Jennings 2005). Releasing live unwanted catch of subtidal benthic invertebrates before handling/processing the commercially targeted species may increase their chances of survival following release back into the water, because they would have been out of the water for less time.

Revill A.S. & Jennings S. (2005) The capacity of benthos release panels to reduce the impacts of beam trawls on benthic communities. *Fisheries Research,* 75, 73–85.

6.56. Modify harvest methods of macroalgae

- We found no studies that evaluated the effects of modifying harvest methods of macroalgae on subtidal benthic invertebrate 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

The commercial harvest of macroalgae (e.g. kelp) can impact subtidal benthic invertebrates through removal of the plant itself, direct physical damage and removal of invertebrates, or through disturbance to the seabed (Pirker 2002; Stagnol *et al*. 2016). The harvest method can be modified in an attempt to prevent such negative impacts. For instance, harvesting macroalgal blades rather than mechanically removing the whole plant can reduce disturbances to the seabed and retain some benefit of macroalgal canopy (Levitt *et al.* 2002). Increasing the time between consecutive harvests can also potentially help reduce the pressure and allow for natural recovery. Similarly, harvesting patches of macroalgae rather than entire areas can potentially allow natural recolonization of subtidal benthic invertebrates from nearby unharvested patches.

Levitt G.J., Anderson R.J., Boothroyd C.J.T. & Kemp F.A. (2002) The effects of kelp harvesting on its regrowth and the understorey benthic community at Danger Point, South Africa, and a new method of harvesting kelp fronds. *South African Journal of Marine Science,* 24, 71–85.

Pirker J.G. (2002) Demography, biomass production and effects of harvesting giant kelp *Macrocystis pyrifera* (Linnaeus) in Southern New Zealand.

Stagnol D., Michel R. & Davoult D. (2016) Unravelling the impact of harvesting pressure on canopy-forming macroalgae. *Marine and Freshwater Research*, 67, 153–161.

7. Threat: Human intrusions and disturbances

Background

Human intrusions and disturbances can originate from a wide array of large-scale activities and impact on subtidal benthic invertebrates. These include residential and industrial development, point discharges, aquaculture, shipping and transportation, energy production and mining, and biological resource use. Interventions related to these threats are described in previous chapters.

Interventions related to protecting, or restoring and recreating habitats following intrusions and disturbances are described in "Habitat protection" and "Habitat restoration and creation", respectively.

Interventions related to human intrusions and disturbances from recreational activities are discussed below. These include activities such as diving or recreational harvesting, which can negatively impact subtidal benthic invertebrates through damage or destruction of habitats, or through direct removal (Hardiman & Burgin 2010; Lloret *et al.* 2008; Saphier & Hoffmann 2005; West 2011). Evidence for interventions related to recreational activities linked with boating has been summarised in "Transportation and service corridors – Shipping lanes".

Hardiman N. & Burgin S. (2010) Recreational impacts on the fauna of Australian coastal marine ecosystems. *Journal of Environmental Management*, 91, 2096–2108.

Lloret J., Zaragoza N., Caballero D. & Riera V. (2008) Impacts of recreational boating on the marine environment of Cap de Creus (Mediterranean Sea). *Ocean & Coastal Management*, 51, 749–754.

Saphier A.D. & Hoffmann T.C. (2005) Forecasting models to quantify three anthropogenic stresses on coral reefs from marine recreation: Anchor damage, diver contact and copper emission from antifouling paint. *Marine Pollution Bulletin*, 51, 590–598.

West R.J. (2011) Impacts of recreational boating activities on the seagrass *Posidonia* in SE Australia. *Wetlands (Australia),* 26, 3–13.

Recreational Activities

7.1. Limit, cease or prohibit access for recreational purposes

- We found no studies that evaluated the effects of limiting, ceasing or prohibiting access for recreational purposes on subtidal benthic invertebrate 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

Recreational activities can impact subtidal benthic invertebrates through species or habitat removal, physical damage, disturbance (Milazzo *et al.* 2002) or pollution (Harriott *et al.* 1997). Boat and other forms of access for recreational purposes could be limited, by restricting access in space and time (duration and occurrence). Permanent or temporary closure could be put in place, or access prohibited through bylaws. Stopping or restricting the access for recreational purposes may help reduce the intensity of the threats associated with boating and recreational activities, such as harvesting, angling, or diving, and potentially allow subtidal benthic invertebrate communities to persist or recover over time.

When restrictions of recreational activities occur in the context of a marine protected area, evidence is summarised under "Habitat protection", including "Habitat protection - Designate a Marine Protected Area and set a no-anchoring zone", "Designate a Marine Protected Area and prohibit the harvest of scallops", "Designate a Marine Protected Area and prohibit the harvest of conch" and "Designate a Marine Protected Area and prohibit the harvest of sea urchins".

Harriott V.J., Davis D. & Banks S.A. (1997) Recreational diving and its impact in marine protected areas in eastern Australia. *Ambio*, 173–179.

Milazzo M., Chemello R., Badalamenti F., Camarda R. & Riggio S. (2002) The Impact of Human Recreational Activities in Marine Protected Areas: What Lessons Should Be Learnt in the Mediterranean Sea? *Marine Ecology,* 23, 280–290.

7.2. Limit, cease or prohibit recreational diving

- We found no studies that evaluated the effects of limiting, ceasing or prohibiting recreational diving on subtidal benthic invertebrate 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

Recreational activities such as diving can impact subtidal benthic invertebrates through physical damage to species and habitats, and disturbance to sediment and organisms (Harriott *et al.* 1997; Luna *et al.* 2009). Recreational diving could be limited, by restricting access in space and time (duration and occurrence) or restricting the type of gears divers are allowed to carry. It could be ceased by setting a permanent or temporary closure, or prohibited through bylaws. This may help reduce the intensity of the threats associated with diving and potentially allow subtidal benthic invertebrate communities to persist or recover over time.

When restrictions of recreational activities occur in the context of a marine protected area, evidence is summarised under "Habitat protection", including "Habitat protection - Designate a Marine Protected Area and set a no-anchoring zone", "Designate a Marine Protected Area and prohibit the harvest of scallops", "Designate a Marine Protected Area and prohibit the harvest of conch" and "Designate a Marine Protected Area and prohibit the harvest of sea urchins". Other evidence for interventions related to recreational boating is summarised under "Threat: Transportation and service corridors – Shipping lanes".

Harriott V.J., Davis D. & Banks S.A. (1997) Recreational diving and its impact in marine protected areas in eastern Australia. *Ambio*, 173–179.

Luna B., Pérez C.V. & Sánchez-Lizaso J.L. (2009) Benthic impacts of recreational divers in a Mediterranean Marine Protected Area. *ICES Journal of Marine Science*, 66, 517–523.

7.3. Limit, cease or prohibit recreational fishing and/or harvesting

- We found no studies that evaluated the effects of limiting, ceasing or prohibiting recreational fishing and/or harvesting on subtidal benthic invertebrate 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

Recreational harvesting (free divers, spear fishers) and fishing can impact subtidal benthic invertebrates through species removal, either intentionally, or unintentionally through accidental unwanted catch (in the case of fishing), physical damage and disturbance (Cooke & Cowx 2006; Milazzo *et al.* 2002). Recreational fishing and harvesting could be limited in one area, by restricting the activity in space and time (limits on duration and occurrence, delimiting allowed areas). It could be ceased by setting a permanent or temporary closure (e.g. seasonal closure), or prohibited through bylaws. This may help reduce the intensity of the threats associated with these activities and potentially allow subtidal benthic invertebrate communities to persist or recover over time.

When restrictions of recreational activities occur in the context of a marine protected area, evidence is summarised under "Habitat protection", including "Habitat protection - Designate a Marine Protected Area and prohibit all types of fishing", "Designate a Marine Protected Area and only allow hook and line fishing", "Designate a Marine Protected Area and prohibit the harvest of scallops", "Designate a Marine Protected Area and prohibit the harvest of conch" and "Designate a Marine Protected Area and prohibit the harvest of sea urchins". Other evidence for interventions related to recreational boating is summarised under "Threat: Transportation and service corridors – Shipping lanes".

Cooke S.J. & Cowx I.G. (2006) Contrasting recreational and commercial fishing: Searching for common issues to promote unified conservation of fisheries resources and aquatic environments. *Biological Conservation,* 128, 93–108.

Milazzo M., Chemello R., Badalamenti F., Camarda R. & Riggio S. (2002) The impact of human recreational activities in Marine Protected Areas: What lessons should be learnt in the Mediterranean Sea? *Marine Ecology,* 23, 280–290.

8. Threat: Invasive and other problematic species, genes and diseases

Background

Non-native, invasive, or other problematic species of animals, plants, algae, and diseases can cause significant adverse consequences to the marine environment and to the local native species (Bax *et al*. 2003; Ruiz *et al*. 1997). There, they can impact on native subtidal benthic invertebrate species through predation, competition for resources (food & space), contamination (for pathogens and diseases), or hybridization (through reproduction) (Bishop *et al*. 2010). The main vectors of introduction of non-native, invasive or problematic species are linked with the development of aquaculture and the intensification of both recreational boating and commercial international trans-ocean transportations (Hewitt & Campbell 2007; Hewitt *et al*. 2004a; Hulme 2009; Molnar *et al.* 2008).

This chapter describes the evidence for interventions designed to prevent, reduce, or mitigate the threat from non-native, invasive and other problematic species. In the marine environment, and particularly in at-risk locations, following the precautionary approach by preventing the introduction of non-native, invasive and problematic species is generally accepted to be the most effective and cost-efficient management option (Hewitt & Campbell 2007; Hewitt *et al*. 2004b; Katsanevakis *et al*. 2013).

Bax N., Williamson A., Aguero M., Gonzalez E. & Geeves W. (2003) Marine invasive alien species: a threat to global biodiversity. *Marine Policy*, 27, 313–323.

Bishop M.J., Krassoi F.R., McPherson R.G., Brown K.R., Summerhayes S.A., Wilkie E.M. & O'Connor W.A. (2010) Change in wild-oyster assemblages of Port Stephens, NSW, Australia, since commencement of non-native Pacific oyster (*Crassostrea gigas*) aquaculture. *Marine and Freshwater Research*, 61, 714–723.

Hewitt C.L. & Campbell M.L. (2007) Mechanisms for the prevention of marine bioinvasions for better biosecurity. *Marine Pollution Bulletin*, 55, 395–401.

Hewitt C.L., Campbell M.L., Thresher R.E., Martin R.B., Boyd S., Cohen B.F., Currie D.R., Gomon M.F., Keough M.J., Lewis J.A., Lockett M.M., Mays N., McArthur M.A., O'Hara T.D., Poore G.C.B., Ross D.J., Storey M., Watson J.E. & Wilson R.S. (2004a) Introduced and cryptogenic species in Port Phillip Bay, Victoria, Australia*. Marine Biology*, 144, 183–202.

Hewitt C.L., Willing J., Bauckham Al., Cassidy A.M., Cox C.M.S., Jones L. & Wotton D.M. (2004b) New Zealand marine biosecurity: Delivering outcomes in a fluid environment*. New Zealand Journal of Marine and Freshwater Research*, 38, 429–438.

Hulme P.E. (2009) Trade, transport and trouble: managing invasive species pathways in an era of globalization. *Journal of Applied Ecology*, 46, 10–18.

Katsanevakis S., Zenetos A., Belchior C. & Cardoso A.C. (2013) Invading European Seas: assessing pathways of introduction of marine aliens. *Ocean & Coastal Management*, 76, 64–74.

Molnar J.L., Gamboa R.L., Revenga C. & Spalding M.D. (2008) Assessing the global threat of invasive species to marine biodiversity. *Frontiers in Ecology and the Environment*, 6, 485–492.

Ruiz G.M., Carlton J.T., Grosholz E.D. & Hines A.H. (1997) Global invasions of marine and estuarine habitats by non-indigenous species: mechanisms, extent, and consequences. *American Zoologist*, 37, 621–632.

Aquaculture

8.1. Use native species instead of non-native species in aquaculture systems

- We found no studies that evaluated the effects of using native species instead of non-native species in aquaculture systems on subtidal benthic invertebrate 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 known to negatively affect local native communities (Arthur *et al,* 2009; Campbell & Hewitt 2008; Molnar *et al*. 2008). Non-native species are commonly used worldwide for aquaculture purposes due to their economic value (for instance, the Pacific oyster *Crassostrea* (also known as *Magallana*) *gigas* is importantly produced in the UK despite not being native). Using native species for aquaculture instead of non-native species removes the risk of introducing non-native species into an area, either intentionally (from at sea culture) or accidentally (as escapees from hatchery facilities; Arechavaia-Lopez *et al*. 2013; Campbell 2009; Campbell 2011). Culturing native species may also reduce pressure on native species populations by displacing the harvest effort from native stocks to aquaculture stocks (Andriahajaina & Hockley 2007).

Andriahajaina & Hockley (2007) The potential of native species aquaculture to achieve conservation objectives: freshwater crayfish in Madagascar. *The International Journal of Biodiversity Science and Management,* 3, 217–222.

Arechavaia-Lopez P., Sanchez-Jerez P., Bayle-Sempere J.T., Uglem I. & Mladineo I. (2013) Reared fish. Farmed escapees and wild fish stockes – a triangle of pathogen transmission of concern to Mediterranean aquaculture management. *Aquaculture Environment Interactions*, 3, 153–161.

Arthur J.R., Bondad-Reantaso M.G., Campbell M.L., Hewitt C.L., Phillips M.J. & Subasinghe R.P. (2009). *Understanding and applying risk analysis in aquaculture: a manual for decision-makers*. FAO Fisheries and Aquaculture Technical Paper No. 519/1. FAO; Rome. 113pp.

Campbell M.L. (2009). An overview of risk assessment in a marine biosecurity context. Chapter 20. 99 353–373 in: Rilov G & Crooks J (eds.). *Biological Invasions in Marine Ecosystems. Ecological, Management, and Geographic Perspectives*. Heidelberg, Germany: Springer

Campbell M.L. (2011) Assessing biosecurity risk associated with the importation of microalgae. *Environmental Research,* 111, 989–998.

Campbell M.L. & Hewitt C.L. (2008) Introduced marine species risk assessment – aquaculture. Pages 121–133 in: M.G. Bondad-Reantaso; J.R. Arthur. & R.P. Subasinghe (eds). *Understanding and applying risk analysis in aquaculture*. FAO Fisheries and Aquaculture Technical Paper. No. 519. Rome, FAO.

Molnar J.L., Gamboa R.L., Revenga C. & Spalding M.D. (2008) Assessing the global threat of invasive species to marine biodiversity. *Frontiers in Ecology and the Environment*, 6, 485–492.

8.2. Implement quarantine to avoid accidental introduction of disease, non-native or problem species

- We found no studies that evaluated the effects of implementing quarantine to avoid accidental introduction of disease, non-native or problem species on subtidal benthic invertebrate 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

The expanding aquaculture industry has led to the accidental introduction of diseases, and non-native and other problematic species into the wild marine environment (Bax *et al*. 2003; Hewitt *et al.* 2004; Manchester & Bullock 2000). There, they can impact on native subtidal benthic invertebrate species through predation, competition for resources (food & space), contamination (for pathogens and diseases), or hybridization (through reproduction) (Bishop *et al*. 2010). This could be potentially avoided through the implementation of quarantine procedures (Campbell 2011; Reise *et al*. 1998).

Evidence for related interventions are summarised under "Threat: Invasive and other problematic species, genes and diseases – Implement regular inspections to avoid accidental introduction of disease or non-native or problem species".

Bax N., Williamson A., Aguero M., Gonzalez E. & Geeves W. (2003) Marine invasive alien species: a threat to global biodiversity. *Marine Policy*, 27, 313–323.

Bishop M.J., Krassoi F.R., McPherson R.G., Brown K.R., Summerhayes S.A., Wilkie E.M. & O'Connor W.A. (2010) Change in wild-oyster assemblages of Port Stephens, NSW, Australia, since commencement of non-native Pacific oyster (*Crassostrea gigas*) aquaculture. *Marine and Freshwater Research*, 61, 714–723.

Hewitt C.L., Campbell M.L., Thresher R.E., Martin R.B., Boyd S., Cohen B.F., Currie D.R., Gomon M.F., Keough M.J., Lewis J.A., Lockett M.M., Mays N., McArthur M.A., O'Hara T.D., Poore G.C.B., Ross D.J., Storey M., Watson J.E. & Wilson R.S. (2004) Introduced and cryptogenic species in Port Phillip Bay, Victoria, Australia. *Marine Biology*, 144, 183–202.

Campbell M.L. (2011) Assessing biosecurity risk associated with the importation of microalgae. *Environmental Research*. 111, 989–998.

Manchester S.J. & Bullock J.M. (2000) The impacts of non-native species on UK biodiversity and the effectiveness of control. *Journal of Applied Ecology*, 37, 845–864.

Reise K., Gollasch S. & Wolff W.J. (1998) Introduced marine species of the North Sea coasts. *Helgoländer Meeresuntersuchungen*, 52, 219.

8.3. Implement regular inspections to avoid accidental introduction of disease or non-native or problem species

- We found no studies that evaluated the effects of implementing regular inspections to avoid accidental introduction of disease, non-native or problem species on subtidal benthic invertebrate 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

The expanding aquaculture industry has led to the accidental introduction of diseases, and non-native and other problematic species into the wild marine environment (Bax *et al*. 2003; Fitridge *et al.* 2012; Hewitt *et al.* 2004; Manchester & Bullock 2000). There, they can impact on native subtidal benthic invertebrate species through predation, competition for resources (food & space), contamination (for pathogens and diseases), or hybridization (through reproduction) (Bishop *et al*. 2010). This could be potentially avoided through the implementation of regular inspections of the facilities (Fitridge *et al.* 2012; Reise *et al*. 1998).

Evidence for related interventions are summarised under "Threat: Invasive and other problematic species, genes and diseases – Implement quarantine to avoid accidental introduction of disease, non-native or problem species".

Bax N., Williamson A., Aguero M., Gonzalez E. & Geeves W. (2003) Marine invasive alien species: a threat to global biodiversity. *Marine Policy*, 27, 313–323.

Bishop M.J., Krassoi F.R., McPherson R.G., Brown K.R., Summerhayes S.A., Wilkie E.M. & O'Connor W.A. (2010) Change in wild-oyster assemblages of Port Stephens, NSW, Australia, since commencement of non-native Pacific oyster (*Crassostrea gigas*) aquaculture. *Marine and Freshwater Research*, 61, 714–723.

Fitridge I., Dempster T., Guenther J. & de Nys R., (2012) The impact and control of biofouling in marine aquaculture: a review. *Biofouling*, 28, 649–669.
Hewitt C.L., Campbell M.L., Thresher R.E., Martin R.B., Boyd S., Cohen B.F., Currie D.R., Gomon M.F., Keough M.J., Lewis J.A., Lockett M.M., Mays N., McArthur M.A., O'Hara T.D., Poore G.C.B., Ross D.J., Storey M., Watson J.E. & Wilson R.S. (2004) Introduced and cryptogenic species in Port Phillip Bay, Victoria, Australia*. Marine Biology*, 144, 183–202.
Manchester S.J. & Bullock J.M. (2000) The impacts of non-native species on UK biodiversity and the effectiveness of control. *Journal of Applied Ecology*, 37, 845–864.
Reise K., Gollasch S. & Wolff W.J. (1998) Introduced marine species of the North Sea coasts. *Helgoländer Meeresuntersuchungen*, 52, 219.

8.4. Use sterile individuals in aquaculture systems using non-native species

- We found no studies that evaluated the effects of using sterile individuals in aquaculture systems using non-native species on subtidal benthic invertebrate 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 commonly used worldwide for aquaculture purposes due to their economic value (for instance, the Pacific oyster *Magallana gigas* is importantly produced in the UK despite not being native). The expanding aquaculture industry has led to the accidental introduction of non-native species into the wild marine environment (Bax *et al*. 2003; Hewitt *et al.* 2004; Manchester & Bullock 2000), where they can impact on native subtidal benthic invertebrate species through hybridization (reproduction). This could be potentially avoided by using only sterile individuals of a non-native species in aquaculture (Thresher *et al*. 2009). However, for this to be a fully effective intervention, advances in polyploidy aquaculture and genetic containment need to occur, given that sterile individuals have been shown to revert over time and that polyploidy can have negative outcomes (e.g. Piferer *et al*. 2009; Zajicek *et al*. 2011).

Bax N., Williamson A., Aguero M., Gonzalez E. & Geeves W. (2003) Marine invasive alien species: a threat to global biodiversity. *Marine Policy*, 27, 313–323.
Hewitt C.L., Campbell M.L., Thresher R.E., Martin R.B., Boyd S., Cohen B.F., Currie D.R., Gomon M.F., Keough M.J., Lewis J.A., Lockett M.M., Mays N., McArthur M.A., O'Hara T.D., Poore G.C.B., Ross D.J., Storey M., Watson J.E. & Wilson R.S. (2004) Introduced and cryptogenic species in Port Phillip Bay, Victoria, Australia*. Marine Biology*, 144, 183–202.
Manchester S.J. & Bullock J.M. (2000) The impacts of non-native species on UK biodiversity and the effectiveness of control. *Journal of Applied Ecology*, 37, 845–864.
Piferer F., Beaumont A., Falguiere J-C., Flajshans M., Haffray P. & Colombo L. (2009) Polyploidy fish and shellfish: Production, biology, and applications to aquaculture for performance improvement and genetic containment. *Aquaculture*, 293, 125–156.
Thresher R., Grewe P., Patil J.G., Whyard S., Templeton C.M., Chaimongol A., Hardy C.M., Hinds L.A. & Dunham R. (2009) Development of repressible sterility to prevent the establishment of feral populations of exotic and genetically modified animals. *Aquaculture*, 290, 104–109.
Zajicek P., Goodwin, A.E., & Weier, T. (2011) Triploid grass carp: Triploid induction, sterility, reversion, and certification. *North American Journal of Fisheries Management,* 31, 614–618.

8.5. Source spat and juveniles from areas or hatcheries not infested with diseases or non-native or problematic species

- We found no studies that evaluated the effects of sourcing spat and juveniles from areas or hatcheries not infested with diseases or non-native or problematic species on subtidal benthic invertebrate 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

The expanding aquaculture industry has led to the accidental introduction of diseases, and non-native and other problematic species into the wild marine environment (Arechavaia-Lopez *et al.* 2013; Bax *et al*. 2003; Campbell & Hewitt 2008; Manchester & Bullock 2000). There, they can impact on native subtidal benthic invertebrate species through predation, competition for resources (food & space), contamination (for pathogens and diseases), or hybridization (through reproduction) (Bishop *et al*. 2010; Fitridge *et al.* 2012). Spat is the name used for very young shellfish, usually mussels or oysters. In aquaculture, spat, as well as juveniles (young adults), can be obtained from hatchery facilities or from natural stocks. Spat and juveniles are then cultured at sea and will grow into marketable adults. Depending on the water quality at the site spat and juveniles are sourced from, individuals can carry diseases and non-native or problematic species, either inside them or on their shell (Brenner *et al.* 2014). The introduction of diseases, non-native and other problematic species to a new environment could be potentially avoided by only selecting spat and juveniles from non-infested areas and hatcheries, for instance accredited or certified facilities.

Arechavaia-Lopez P., Sanchez-Jerez P., Bayle-Sempere J.T., Uglem I. & Mladineo I. (2013) Reared fish. Farmed escapees and wild fish stockes – a triangle of pathogen transmission of concern to Mediterranean aquaculture management. *Aquaculture Environment Interactions*, 3, 153–161.

Bax N., Williamson A., Aguero M., Gonzalez E. & Geeves W. (2003) Marine invasive alien species: a threat to global biodiversity. *Marine Policy*, 27, 313–323.

Bishop M.J., Krassoi F.R., McPherson R.G., Brown K.R., Summerhayes S.A., Wilkie E.M. & O'Connor W.A. (2010) Change in wild-oyster assemblages of Port Stephens, NSW, Australia, since commencement of non-native Pacific oyster (*Crassostrea gigas*) aquaculture. *Marine and Freshwater Research*, 61, 714–723.

Brenner M., Fraser D., Van Nieuwenhove K., O'Beirn F., Buck B.H., Mazurié J., Thorarinsdottir G., Dolmer P., Sanchez-Mata A., Strand O. & Flimlin G. (2014) Bivalve aquaculture transfers in Atlantic Europe. Part B: environmental impacts of transfer activities. *Ocean & Coastal Management*, 89, 139–146.

Campbell M.L. & Hewitt C.L. (2008) Introduced marine species risk assessment – aquaculture. Pages 121–133 in: M.G. Bondad-Reantaso; J.R. Arthur. & R.P. Subasinghe (eds). *Understanding and Applying Risk Analysis in Aquaculture*. FAO Fisheries and Aquaculture Technical Paper. No. 519. Rome, FAO.

Fitridge I., Dempster T., Guenther J. & de Nys R., (2012) The impact and control of biofouling in marine aquaculture: a review. *Biofouling*, 28, 649–669.

Manchester S.J. & Bullock J.M. (2000) The impacts of non-native species on UK biodiversity and the effectiveness of control. *Journal of Applied Ecology*, 37, 845–864.

8.6. Import spat and/or eggs to aquaculture facilities rather than juveniles and adults to reduce the risk of introducing hitchhiking species

- We found no studies that evaluated the effects of importing spat and/or eggs to aquaculture facilities rather than juveniles and adults to reduce the risk of introducing hitchhiking species on subtidal benthic invertebrate 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

The expanding aquaculture industry has led to the accidental introduction of diseases, and non-native and other problematic species into the wild marine environment (Arechavaia-Lopez *et al.* 2013; Bax *et al*. 2003; Campbell & Hewitt 2008; Manchester & Bullock 2000). There, they can impact on native subtidal benthic invertebrate species through predation, competition for resources (food & space), contamination (for pathogens and diseases), or hybridization (through reproduction) (Bishop *et al*. 2010; Fitridge *et al.* 2012). In aquaculture, importing juveniles (young adults) or adults into farming facilities is a common practice, but can lead to the accidental release of non-native or problematic species that hitchhiked on/in them during transport. By importing spat (very young shellfish, usually mussels or oysters) and/or eggs instead, the risk of transporting and releasing these hitchhikers can potentially be reduced.

Arechavaia-Lopez P., Sanchez-Jerez P., Bayle-Sempere J.T., Uglem I. & Mladineo I. (2013) Reared fish. Farmed escapees and wild fish stockes – a triangle of pathogen transmission of concern to Mediterranean aquaculture management. *Aquaculture Environment Interactions*, 3, 153–161.

Bax N., Williamson A., Aguero M., Gonzalez E. & Geeves W. (2003) Marine invasive alien species: a threat to global biodiversity. *Marine Policy*, 27, 313–323.

Bishop M.J., Krassoi F.R., McPherson R.G., Brown K.R., Summerhayes S.A., Wilkie E.M. & O'Connor W.A. (2010) Change in wild-oyster assemblages of Port Stephens, NSW, Australia, since commencement of non-native Pacific oyster (*Crassostrea gigas*) aquaculture. *Marine and Freshwater Research*, 61, 714–723.

Campbell M.L. & Hewitt C.L. (2008) Introduced marine species risk assessment – aquaculture. Pages 121–133 in: M.G. Bondad-Reantaso; J.R. Arthur. & R.P. Subasinghe (eds). *Understanding and Applying Risk Analysis in Aquaculture*. FAO Fisheries and Aquaculture Technical Paper. No. 519. Rome, FAO.

Fitridge I., Dempster T., Guenther J. & de Nys R., (2012) The impact and control of biofouling in marine aquaculture: a review. *Biofouling*, 28, 649–669.

Manchester S.J. & Bullock J.M. (2000) The impacts of non-native species on UK biodiversity and the effectiveness of control. *Journal of Applied Ecology*, 37, 845–864.

8.7. Reduce and/or eradicate aquaculture escapees in the wild

- We found no studies that evaluated the effects of reducing and/or eradicating aquaculture escapees in the wild on subtidal benthic invertebrate 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

The expanding aquaculture industry has led to the accidental introduction of non-native and other problematic species into the wild marine environment, referred to as *escapees* (Arechavaia-Lopez *et al*. 2013; Bax *et al*. 2003; Manchester & Bullock, 2000). There, they can impact on native subtidal benthic invertebrate species through predation, competition for resources (food & space), or hybridization (through reproduction) (Bishop *et al*. 2010). Managing the spread of escapees, either by reducing their populations or trying to eradicate them when feasible (Herbert *et al*. 2016), can potentially reduce the threat level and associated risks on native subtidal benthic invertebrate species. Additionally, revising the existing spacing systems for farms, using innovative siting systems, and improving cage technologies and operational routines, could be effective means of reducing the likelihood of escapees (Arechavaia-Lopez *et al*. 2013)

Arechavaia-Lopez P., Sanchez-Jerez P., Bayle-Sempere J.T., Uglem I. & Mladineo I. (2013) Reared fish. Farmed escapees and wild fish stockes – a triangle of pathogen transmission of concern to Mediterranean aquaculture management. *Aquaculture Environment Interactions*, 3, 153–161.

Bax N., Williamson A., Aguero M., Gonzalez E. & Geeves W. (2003) Marine invasive alien species: a threat to global biodiversity. *Marine Policy*, 27, 313–323.

Bishop M.J., Krassoi F.R., McPherson R.G., Brown K.R., Summerhayes S.A., Wilkie E.M. & O'Connor W.A. (2010) Change in wild-oyster assemblages of Port Stephens, NSW, Australia, since commencement of non-native Pacific oyster (*Crassostrea gigas*) aquaculture. *Marine and Freshwater Research*, 61, 714–723.

Herbert R.J., Humphreys J., Davies C.J., Roberts C., Fletcher S. & Crowe T.P. (2016) Ecological impacts of non-native Pacific oysters (*Crassostrea gigas*) and management measures for protected areas in Europe. *Biodiversity and Conservation*, 25, 2835–2865.

Manchester S.J. & Bullock J.M. (2000) The impacts of non-native species on UK biodiversity and the effectiveness of control. *Journal of Applied Ecology*, 37, 845–864.

8.8. Prevent the attachment of biofouling organisms/species in aquaculture

- We found no studies that evaluated the effects of preventing the attachment of biofouling organisms/species in aquaculture on subtidal benthic invertebrate 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

While aquaculture facilities can be partly located on land (hatcheries), most of it occurs at sea, in cages, pens, bags, or ropes exposed to the marine environment. They represent hard structures onto which organisms can attach and grow – those organisms are known as the *biofouling community*. Non-native, invasive and other problematic species can be part of this biofouling community (Fitridge *et al*. 2012) and use aquaculture structures as "*stepping stones*" to spread and reach new areas to colonize (Ruiz *et al*. 1997). They can impact on native subtidal benthic invertebrate species through predation, competition for resources (food & space), contamination (for pathogens and diseases), or hybridization (through reproduction) (Bishop *et al*. 2010). Preventing the attachment of biofouling organisms can potentially help reduce the risks that invasive, non-native and other problematic biofouling species pose to subtidal benthic invertebrates. Non-fouling material, anti-fouling paints or coatings can be used for aquaculture infrastructures to prevent attachment (Fitridge *et al*. 2012).

Evidence for other interventions related to biofouling are summarised under "Threat: Invasive and other problematic species, genes and diseases – Remove biofouling organisms/species in aquaculture", "Clean anthropogenic platforms, structures or equipment", "Use antifouling coatings on the surfaces of vessels and anthropogenic structures", "Use non-toxic antifouling coatings on surfaces" and "Restrict the use of tributyltin or other toxic antifouling coatings".

Bishop M.J., Krassoi F.R., McPherson R.G., Brown K.R., Summerhayes S.A., Wilkie E.M. & O'Connor W.A. (2010) Change in wild-oyster assemblages of Port Stephens, NSW, Australia, since commencement of non-native Pacific oyster (*Crassostrea gigas*) aquaculture. *Marine and Freshwater Research*, 61, 714–723.

Fitridge I., Dempster T., Guenther J. & de Nys R., (2012) The impact and control of biofouling in marine aquaculture: a review. *Biofouling*, 28, 649–669.

Ruiz G.M., Carlton J.T., Grosholz E.D. & Hines A.H. (1997) Global invasions of marine and estuarine habitats by non-indigenous species: mechanisms, extent, and consequences. *American Zoologist*, 37, 621–632.

8.9. Remove biofouling organisms/species in aquaculture

- We found no studies that evaluated the effects of removing biofouling organisms/species in aquaculture on subtidal benthic invertebrate 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

While aquaculture facilities can be partly located on land (hatcheries), most of it occurs at sea, in cages, pens, bags, or ropes exposed to the marine environment. They represent hard structures onto which organisms can attach and grow – known as the *biofouling community*. Non-native, invasive and other problematic species can be part of this biofouling community (Fitridge *et al*. 2012) and use aquaculture structures as "*stepping stones*" to spread and reach new areas to colonize (Ruiz *et al*. 1997). Non-native, invasive and problematic species can impact on native subtidal benthic invertebrate species through predation, competition for resources (food & space), contamination (for pathogens and diseases), or hybridization (through reproduction) (Bishop *et al*. 2010). Regularly removing biofouling organisms can potentially help reduce the risks that invasive, non-native and other problematic biofouling species pose to subtidal benthic invertebrates. Biofouling species can be manually or mechanically removed by introducing biological agents (such as a predatory species), or by undertaking regular cleaning of infrastructures (Fitridge *et al*. 2012). When removing biofouling care must be taken to ensure that biofouled marine debris is not created, as these can float to new destinations where these biofouling species can spread (Campbell *et al.* 2017).

Evidence for other interventions related to biofouling are summarised under "Threat: Invasive and other problematic species, genes and diseases – Prevent the attachment of biofouling organisms/species in aquaculture", "Clean anthropogenic platforms, structures or equipment", "Use antifouling coatings on the surfaces of vessels and anthropogenic structures", "Use non-toxic antifouling coatings on surfaces" and "Restrict the use of tributyltin or other toxic antifouling coatings".

Bishop M.J., Krassoi F.R., McPherson R.G., Brown K.R., Summerhayes S.A., Wilkie E.M. & O'Connor W.A. (2010) Change in wild-oyster assemblages of Port Stephens, NSW, Australia, since commencement of non-native Pacific oyster (*Crassostrea gigas*) aquaculture. *Marine and Freshwater Research*, 61, 714–723.

Campbell M.L., King S., Heppenstall L.D., van Gool E., Martin R. & Hewitt C.L. (2017) Aquaculture and urban marine structures facilitate native and non-indigenous species transfer through generation and accumulation of marine debris. *Marine Pollution Bulletin*, 123, 304–312.

Fitridge I., Dempster T., Guenther J. & de Nys R., (2012) The impact and control of biofouling in marine aquaculture: a review. *Biofouling*, 28, 649–669.

Ruiz G.M., Carlton J.T., Grosholz E.D. & Hines A.H. (1997) Global invasions of marine and estuarine habitats by non-indigenous species: mechanisms, extent, and consequences. *American Zoologist*, 37, 621–632.

Shipping, transportation and anthropogenic structures

8.10. Limit, cease or prohibit ballast water exchange in specific areas

• We found no studies that evaluated the effects of limiting, ceasing or prohibiting ballast water exchange in specific areas on subtidal benthic invertebrate 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, invasive and other problematic species can impact on native subtidal benthic invertebrate species through predation, competition for resources (food & space), contamination (for pathogens and diseases), or hybridization (through reproduction) (Molnar *et al*. 2008; Bishop *et al*. 2010). Ballasting is the process by which sea water (ballast water) is taken in and out of the ship when the ship is at port or at sea. Ballast water can therefore contain species from one location taken-up during water intake, which are then accidentally released in a new environment during de-ballasting (water release). Ballast water is one of the major processes of introduction of non-native, invasive and problematic species (Barry *et al.* 2008; Hewitt 2003; Hewitt *et al.* 2004; Molnar *et al.* 2008). Limiting, ceasing, or prohibiting ballast water exchange in specific areas may potentially help prevent the introduction, the establishment and the spread of non-native species and potentially invasive species. Limiting introduction, establishment and spread of such species could be achieved by setting new zone boundaries where ballasting is allowed, setting timing where risk is reduced, or setting limits on the number of ships allowed to ballast at any given time.

Related evidence is summarised under "Threat: Invasive and other problematic species, genes and diseases – Treat ballast water before exchange".

Barry S.C., Hayes K.R., Hewitt C.L., Behrens H.L., Dragsund E. & Bakke S.M. (2008) Ballast water risk assessment: principles, processes and methods. *ICES Journal of Marine Science*, 65, 121–131.

Bishop M.J., Krassoi F.R., McPherson R.G., Brown K.R., Summerhayes S.A., Wilkie E.M. & O'Connor W.A. (2010) Change in wild-oyster assemblages of Port Stephens, NSW, Australia, since commencement of non-native Pacific oyster (*Crassostrea gigas*) aquaculture. *Marine and Freshwater Research*, 61, 714–723.

Hewitt C.L. (2003). Marine biosecurity issues in the world oceans: global activities and Australian directions. *Ocean Yearbook*, 17, 193–212.

Hewitt C.L., Campbell M.L., Thresher R.E., Martin R.B., Boyd S., Cohen B.F., Currie D.R., Gomon M.F., Keough M.J., Lewis J.A., Lockett M.M., Mays N., McArthur M.A., O'Hara T.D., Poore G.C.B., Ross D.J., Storey M., Watson J.E. & Wilson R.S. (2004) Introduced and cryptogenic species in Port Phillip Bay, Victoria, Australia*. Marine Biology*, 144, 183–202. Molnar J.L., Gamboa R.L., Revenga C. & Spalding M.D. (2008) Assessing the global threat of invasive species to marine biodiversity. *Frontiers in Ecology and the Environment*, 6, 485–492.

8.11. Treat ballast water before exchange

• We found no studies that evaluated the effects of treating ballast water before exchange on subtidal benthic invertebrate 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, invasive and other problematic species can impact on native subtidal benthic invertebrate species through predation, competition for resources (food & space), contamination (for pathogens and diseases), or hybridization (through reproduction) (Molnar *et al*. 2008; Bishop *et al*. 2010). Ballasting is the process by which sea water (ballast water) is taken in and out of the ship when the ship is at port or at sea. Ballast water can therefore contain species from one location taken-up during water intake, which are then accidentally released in a new environment during de-ballasting (water release). Ballast water is one of the major processes of introduction of non-native, invasive and problematic species (Barry *et al.* 2008; Hewitt 2003; Hewitt *et al.* 2004; Molnar *et al.* 2008). Treating ballast waters before exchange can potentially eliminate most or all risks of accidental introduction of non-native, invasive or other problematic species (Reise *et al*. 1998). Treating ballast waters can be done through either mechanical–physical or chemical processes, for instance using high-performance filters, oxidizing or disinfecting chemicals, or Ultra-Violet radiations (Werschkun *et al*. 2014).

Related evidence is summarised under "Threat: Invasive and other problematic species, genes and diseases – Limit, cease or prohibit ballast water exchange in specific areas".

Barry S.C., Hayes K.R., Hewitt C.L., Behrens H.L., Dragsund E. & Bakke S.M. (2008) Ballast water risk assessment: principles, processes and methods. *ICES Journal of Marine Science*, 65, 121–131.

Bishop M.J., Krassoi F.R., McPherson R.G., Brown K.R., Summerhayes S.A., Wilkie E.M. & O'Connor W.A. (2010) Change in wild-oyster assemblages of Port Stephens, NSW, Australia, since commencement of non-native Pacific oyster (*Crassostrea gigas*) aquaculture. *Marine and Freshwater Research*, 61, 714–723.

Hewitt C.L. (2003). Marine biosecurity issues in the world oceans: global activities and Australian directions. *Ocean Yearbook*, 17, 193–212.

Hewitt C.L., Campbell M.L., Thresher R.E., Martin R.B., Boyd S., Cohen B.F., Currie D.R., Gomon M.F., Keough M.J., Lewis J.A., Lockett M.M., Mays N., McArthur M.A., O'Hara T.D., Poore G.C.B., Ross D.J., Storey M., Watson J.E. & Wilson R.S. (2004) Introduced and cryptogenic species in Port Phillip Bay, Victoria, Australia. *Marine Biology*, 144, 183–202. Molnar J.L., Gamboa R.L., Revenga C. & Spalding M.D. (2008) Assessing the global threat of invasive species to marine biodiversity. *Frontiers in Ecology and the Environment*, 6, 485–492.

Reise K., Gollasch S. & Wolff W.J. (1998) Introduced marine species of the North Sea coasts. *Helgoländer Meeresuntersuchungen*, 52, 219.

Werschkun B., Banerji S., Basurko O.C., David M., Fuhr F., Gollasch S., Grummt T., Haarich M., Jha A.N., Kacan S. & Kehrer A. (2014) Emerging risks from ballast water treatment: The run-up to the International Ballast Water Management Convention. *Chemosphere*, 112, 256–266.

8.12. Clean the hull, anchor and chain of commercial and recreational vessels

- We found no studies that evaluated the effects of cleaning the hull, anchor and chain of commercial and recreational vessels on subtidal benthic invertebrate 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, invasive and other problematic species can impact on native subtidal benthic invertebrate species through predation, competition for resources (food & space), contamination (for pathogens and diseases), or hybridization (through reproduction) (Molnar *et al*. 2008; Bishop *et al*. 2010). Commercial vessels are major means of trans-oceanic transport of non-native species, while recreational boating is known to facilitate the local spread once in a new environment (Campbell & Hewitt 1999; Clarke *et al.* 2011; Hewitt *et al.* 2004). Non-native species can become attached to the hard surfaces of ships and boats, including the hull, anchor, and chain, and be accidentally transported from one location to another (Campbell & Hewitt 1999; Hewitt *et al.* 2004). Regular cleaning of hulls, anchors and chains can potentially reduce the risk of introduction to new location, and as such reduce the risk non-native species pose to native subtidal benthic invertebrates.

Evidence related to the cleaning of surfaces is summarised under "Threat: Invasive and other problematic species, genes and diseases – Clean anthropogenic platforms, structures or equipment".

Bishop M.J., Krassoi F.R., McPherson R.G., Brown K.R., Summerhayes S.A., Wilkie E.M. & O'Connor W.A. (2010) Change in wild-oyster assemblages of Port Stephens, NSW, Australia, since commencement of non-native Pacific oyster (*Crassostrea gigas*) aquaculture. *Marine and Freshwater Research*, 61, 714–723.

Campbell M.L. & Hewitt C.L. (1999) Vectors, shipping and trade. Pages 45–60 in: Hewitt C L, Campbell ML, Thresher RE, Martin RB (eds.). *The Introduced Species of Port Phillip Bay, Victoria*. Centre for Research on Introduced Marine Pests, CSIRO Marine Research, Hobart.

Clarke Murray C., Pakhomov E.A. & Therriault T.W. (2011) Recreational boating: a large unregulated vector transporting marine invasive species. *Diversity and Distributions*, 17, 1161–1172.

Hewitt C.L., Campbell M.L., Thresher R.E., Martin R.B., Boyd S., Cohen B.F., Currie D.R., Gomon M.F., Keough M.J., Lewis J.A., Lockett M.M., Mays N., McArthur M.A., O'Hara T.D., Poore G.C.B., Ross D.J., Storey M., Watson J.E. & Wilson R.S. (2004) Introduced and cryptogenic species in Port Phillip Bay, Victoria, Australia. *Marine Biology*, 144, 183–202. Molnar J.L., Gamboa R.L., Revenga C. & Spalding M.D. (2008) Assessing the global threat of invasive species to marine biodiversity. *Frontiers in Ecology and the Environment*, 6, 485–492.

8.13. Clean anthropogenic platforms, structures or equipment

- We found no studies that evaluated the effects of cleaning anthropogenic platforms, structures or equipment on subtidal benthic invertebrate 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, invasive and other problematic species can impact on native subtidal benthic invertebrate species through predation, competition for resources (food & space), contamination (for pathogens and diseases), or hybridization (through reproduction) (Molnar *et al*. 2008; Bishop *et al*. 2010). Non-native species can become attached to the hard surfaces of anthropogenic structures, such as oil rigs, wind farms, pontoons, or buoys, which then act as "*stepping stones*" for their introduction into new environments (Adams *et al*. 2014; Bulleri & Airoldi 2005; Mineur *et al*. 2012). Regular cleaning of these structures can potentially reduce the risk of introduction to new location, and as such reduce the risk non-native species pose to native subtidal benthic invertebrates.

Evidence related to the cleaning of surfaces is summarised under "Threat: Invasive and other problematic species, genes and diseases – Clean the hull, anchor and chain of commercial and recreational vessels".

Adams T.P., Miller R.G., Aleynik D. & Burrows M.T. (2014) Offshore marine renewable energy devices as stepping stones across biogeographical boundaries. *Journal of Applied Ecology*, 51, 330–338.

Bulleri F. & Airoldi L. (2005) Artificial marine structures facilitate the spread of a non-indigenous green alga, *Codium fragile* ssp. *tomentosoides*, in the north Adriatic Sea. *Journal of Applied Ecology*, 42, 1063–1072.

Bishop M.J., Krassoi F.R., McPherson R.G., Brown K.R., Summerhayes S.A., Wilkie E.M. & O'Connor W.A. (2010) Change in wild-oyster assemblages of Port Stephens, NSW, Australia, since commencement of non-native Pacific oyster (*Crassostrea gigas*) aquaculture. *Marine and Freshwater Research*, 61, 714–723.

Mineur F., Cook E.J., Minchin D., Bohn K., MacLeod A. & Maggs C.A. (2012) Changing coasts: Marine aliens and artificial structures. Pages 198–243 in: *Oceanography and Marine Biology*. CRC Press.

Molnar J.L., Gamboa R.L., Revenga C. & Spalding M.D. (2008) Assessing the global threat of invasive species to marine biodiversity. *Frontiers in Ecology and the Environment*, 6, 485–492.

8.14. Use antifouling coatings on the surfaces of vessels and anthropogenic structures

- We found no studies that evaluated the effects of using antifouling coatings on the surfaces of vessels and anthropogenic structures on subtidal benthic invertebrate 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, invasive and other problematic species can impact on native subtidal benthic invertebrate species through predation, competition for resources (food & space), contamination (for pathogens and diseases), or hybridization (through reproduction) (Molnar *et al*. 2008; Bishop *et al*. 2010). Non-native species can become attached to the hard surfaces of vessels and anthropogenic structures, such as ship hull, anchors and chains, oil rigs, wind farms, pontoons, or buoys, which then act as "*stepping stones*" for their introduction into new environments (Adams *et al*. 2014; Bulleri & Airoldi 2005; Mineur *et al*. 2012). Using antifouling coating can potentially prevent the attachment of non-native (and native) species, hence reduce the risk of introduction to new location and the risk non-native species pose to native subtidal benthic invertebrates.

Evidence for other interventions related to antifouling coatings are summarised under "Threat: Invasive and other problematic species, genes and diseases – Use non-toxic antifouling coatings on surfaces" and "Restrict the use of tributyltin or other toxic antifouling coatings".

Adams T.P., Miller R.G., Aleynik D. & Burrows M.T. (2014) Offshore marine renewable energy devices as stepping stones across biogeographical boundaries. *Journal of Applied Ecology*, 51, 330–338.

Bulleri F. & Airoldi L. (2005) Artificial marine structures facilitate the spread of a non-indigenous green alga, *Codium fragile* ssp. *tomentosoides*, in the north Adriatic Sea. *Journal of Applied Ecology*, 42, 1063–1072.

Bishop M.J., Krassoi F.R., McPherson R.G., Brown K.R., Summerhayes S.A., Wilkie E.M. & O'Connor W.A. (2010) Change in wild-oyster assemblages of Port Stephens, NSW, Australia, since commencement of non-native Pacific oyster (*Crassostrea gigas*) aquaculture. *Marine and Freshwater Research*, 61, 714–723.

Mineur F., Cook E.J., Minchin D., Bohn K., MacLeod A. & Maggs C.A. (2012) Changing coasts: Marine aliens and artificial structures. Pages 198–243 in: *Oceanography and Marine Biology*. CRC Press.

Molnar J.L., Gamboa R.L., Revenga C. & Spalding M.D. (2008) Assessing the global threat of invasive species to marine biodiversity. *Frontiers in Ecology and the Environment*, 6, 485–492.

Other

8.15. Limit, cease or prohibit the sale and/or transportation of commercial non-native species

• We found no studies that evaluated the effects of limiting, ceasing or prohibiting the sale and/or transportation of commercial non-native species on subtidal benthic invertebrate 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, invasive and other problematic species can be introduced through trade, for instance by purchasing and using non-native live bait for angling (Kilian *et al.* 2012) or the importation of microalgae for aquaculture feed (Campbell 2011). Restricting or ceasing the sale and transportation of such species, as well as putting additional management controls in place such as disposing of live non-native species on land rather than at sea, may potentially help reduce the risk they pose to subtidal benthic invertebrates, through a reduction in introduction and spread.

Campbell M.L. (2011) Assessing biosecurity risk associated with the importation of microalgae. *Environmental Research*, 111, 989–998.

Kilian J.V., Klauda R.J., Widman S., Kashiwagi M., Bourquin R., Weglein S. & Schuster J. (2012) An assessment of a bait industry and angler behavior as a vector of invasive species. *Biological Invasions,* 14, 1469–1481.

8.16. Genetically modify non-native, invasive or other problematic species

• We found no studies that evaluated the effects of genetically modifying non-native, invasive or other problematic species on subtidal benthic invertebrate 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, invasive and other problematic species can impact on native subtidal benthic invertebrate species through predation, competition for resources (food & space), contamination (for pathogens and diseases), or hybridization (through reproduction) (Molnar *et al*. 2008; Bishop *et al*. 2010). Some individuals of non-native, invasive or other problematic species could be genetically modified (for instance by introducing Trojan sex chromosomes) and introduced to the population to reduce their environmental tolerance, fitness or reproductive capacity (Allendorf & Lundquist 2003; Cotton & Wedekind 2007). This can potentially reduce their ability to hybridize with native species, but also reduce their population over time and with it the threats they pose to native species.

Allendorf F.W. & Lundquist L.L. (2003) Introduction: population biology, evolution, and control of invasive species. *Conservation Biology*, 17, 24–30.

Bishop M.J., Krassoi F.R., McPherson R.G., Brown K.R., Summerhayes S.A., Wilkie E.M. & O'Connor W.A. (2010) Change in wild-oyster assemblages of Port Stephens, NSW, Australia, since commencement of non-native Pacific oyster (*Crassostrea gigas*) aquaculture. *Marine and Freshwater Research*, 61, 714–723.

Cotton S. & Wedekind C. (2007) Control of introduced species using Trojan sex chromosomes. *Trends in Ecology & Evolution*, 22, 441–443.

Molnar J.L., Gamboa R.L., Revenga C. & Spalding M.D. (2008) Assessing the global threat of invasive species to marine biodiversity. *Frontiers in Ecology and the Environment*, 6, 485–492.

8.17. Use biocides or other chemicals to control non-native, invasive or other problematic species

- We found no studies that evaluated the effects of using biocides or other chemicals to control non-native, invasive or other problematic species on subtidal benthic invertebrate 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, invasive and other problematic species can impact on native subtidal benthic invertebrate species through predation, competition for resources (food & space), contamination (for pathogens and diseases), or hybridization (through reproduction) (Molnar *et al.* 2008; Bishop *et al.* 2010). Biocides are chemical substances or microorganisms used with the intention of controlling a problematic species (Fitridge *et al.* 2012; Thresher & Kuris 2004). Using biocides or other chemicals, such as chemical inhibitors, to reduce or control the population of non-native, invasive or other problematic species can lower the risk they pose to subtidal benthic invertebrates.

Bishop M.J., Krassoi F.R., McPherson R.G., Brown K.R., Summerhayes S.A., Wilkie E.M. & O'Connor W.A. (2010) Change in wild-oyster assemblages of Port Stephens, NSW, Australia, since commencement of non-native Pacific oyster (*Crassostrea gigas*) aquaculture. *Marine and Freshwater Research*, 61, 714–723.

Fitridge I., Dempster T., Guenther J. & de Nys R., (2012) The impact and control of biofouling in marine aquaculture: a review. *Biofouling*, 28, 649–669.

Molnar J.L., Gamboa R.L., Revenga C. & Spalding M.D. (2008) Assessing the global threat of invasive species to marine biodiversity. *Frontiers in Ecology and the Environment*, 6, 485–492.

Thresher R., Grewe P., Patil J.G., Whyard S., Templeton C.M., Chaimongol A., Hardy C.M., Hinds L.A. & Dunham R. (2009) Development of repressible sterility to prevent the establishment of feral populations of exotic and genetically modified animals. *Aquaculture*, 290, 104–109.

8.18. Use biological control to manage non-native, invasive or other problematic species populations

• We found no studies that evaluated the effects of using biological control to manage non-native, invasive or other problematic species populations on subtidal benthic invertebrate 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, invasive and other problematic species can impact on native subtidal benthic invertebrate species through predation, competition for resources (food & space), contamination (for pathogens and diseases), or hybridization (through reproduction) (Molnar *et al*. 2008; Bishop *et al*. 2010). Biological controls can be used to try to reduce the population of non-native, invasive or other problematic species (Fitridge *et al.* 2012; Thresher & Kuris 2004). Forms of biological controls include the release of native or non-native predators, parasites, or diseases likely to affect specific non-native, invasive or other problematic species. It should be kept in mind that using native species as biological controls is always a preferred safer option than using non-native ones (Thresher & Kuris 2004).

Bishop M.J., Krassoi F.R., McPherson R.G., Brown K.R., Summerhayes S.A., Wilkie E.M. & O'Connor W.A. (2010) Change in wild-oyster assemblages of Port Stephens, NSW, Australia, since commencement of non-native Pacific oyster (*Crassostrea gigas*) aquaculture. *Marine and Freshwater Research*, 61, 714–723.

Fitridge I., Dempster T., Guenther J. & de Nys R., (2012) The impact and control of biofouling in marine aquaculture: a review. *Biofouling*, 28, 649–669.

Molnar J.L., Gamboa R.L., Revenga C. & Spalding M.D. (2008) Assessing the global threat of invasive species to marine biodiversity. *Frontiers in Ecology and the Environment*, 6, 485–492.

Thresher R., Grewe P., Patil J.G., Whyard S., Templeton C.M., Chaimongol A., Hardy C.M., Hinds L.A. & Dunham R. (2009) Development of repressible sterility to prevent the establishment of feral populations of exotic and genetically modified animals. *Aquaculture*, 290, 104–109.

8.19. Remove or capture non-native, invasive or other problematic species

- **One study** examined the effects of removing or capturing non-native, invasive or other problematic species on subtidal benthic invertebrates. The study was in the South Atlantic Ocean[1] (Brazil).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Cnidarian abundance (1 study):** One replicated, controlled, before-and-after study in the southwest Atlantic[1] found that, regardless of the method used, removing invasive corals reduced the cover of native zoanthids.
- **Sponge abundance (1 study):** One replicated, controlled, before-and-after study in the southwest Atlantic[1] found that the effect of removing invasive corals on the cover of native sponges varied with the removal method used.

Background

Non-native, invasive and other problematic species can impact on native subtidal benthic invertebrate species through predation, competition for resources (food & space), contamination (from pathogens and diseases), or hybridization (through reproduction) (Molnar *et al*. 2008; Bishop *et al*. 2010). Physical removal can be used to attempt to control populations of non-native, invasive or other problematic species (Hewitt *et al*. 2005; Thresher & Kuris 2004). Physical removal can be achieved by using tools (Piazzi & Ceccherelli 2006), manually (Hewitt *et al*. 2005; Wright *et al.* 2005), or through capture (Calderwood *et al*. 2015). Capture can be carried out for instance by using sex pheromones or baited traps to attract the target species (Calderwood *et al*. 2015).

For example, problematic overgrazing sea urchins, for instance due to range extension, can cause a shift from kelp forests to barren areas (Wright *et al.* 2005). Their removal can allow for the kelp forest to recover over time (Wright *et al.* 2005), and in turn help subtidal benthic invertebrates associated with kelp forest recover as well.

Bishop M.J., Krassoi F.R., McPherson R.G., Brown K.R., Summerhayes S.A., Wilkie E.M. & O'Connor W.A. (2010) Change in wild-oyster assemblages of Port Stephens, NSW, Australia, since commencement of non-native Pacific oyster (*Crassostrea gigas*) aquaculture. *Marine and Freshwater Research*, 61, 714–723.

Calderwood J., O'Connor N.E. & Roberts D. (2015) Effects of baited crab pots on cultivated mussel (*Mytilus edulis*) survival rates. *ICES Journal of Marine Science*, 72, 1802–1810.

Hewitt C.L., Campbell M.L., McEnnulty F., Moore M.M., Murfet N.B., Robertson B. & Schaffelke B. (2005) Efficacy of physical removal of a marine pest: the introduced kelp *Undaria pinnatifida* in a Tasmanian Marine Reserve. *Biological Invasions*, 7, 251–263.

Molnar J.L., Gamboa R.L., Revenga C. & Spalding M.D. (2008) Assessing the global threat of invasive species to marine biodiversity. *Frontiers in Ecology and the Environment*, 6, 485–492.

Thresher R., Grewe P., Patil J.G., Whyard S., Templeton C.M., Chaimongol A., Hardy C.M., Hinds L.A. & Dunham R. (2009) Development of repressible sterility to prevent the establishment of feral populations of exotic and genetically modified animals. *Aquaculture*, 290, 104–109.

Wright J.T., Dworjanyn S.A., Rogers C.N., Steinberg P.D., Williamson J.E. & Poore A.G. (2005) Density-dependent sea urchin grazing: differential removal of species, changes in community composition and alternative community states. *Marine Ecology Progress Series*, 298, 143–156.

A replicated, controlled, before-and-after study in 2004–2006 of 20 plots in one rocky reef area of the southwest Atlantic, Brazil (1) found that after a year, the effect of removing the invasive corals *Tubastraea coccinea* and *Tubastraea tagusensis* on the cover of native zoanthid *Palythoa caribaeorum* and native sponges varied with the removal method used. Sponge cover was greater in plots where multiple removals of invasive corals occurred (35%), and lower in plots where removal occurred once (15%), where the whole seabed community was removed once (21%), and where no removal occurred (17%). Zoanthid cover was lower in the single-removal plots (10%) compared to the no-removal plots (22%), while community-removal plots were never recolonised (0% cover after a year). Zoanthids were absent from the multiple-removal plots before removal and did not colonise over time. After a year, invasive corals had recolonised all removal plots (single-removal: 14%; multiple-removal: 3%; community-removal: 14%; no removal: 27%). The two corals invaded the reef approximately 20 years prior. Twenty 0.16 m^2 plots, all with ≥20% cover of invasive corals were selected. Four treatments were used (5 plots/treatment): a single removal of invasive corals (December 2004), multiple removals of invasive corals, a single removal of the whole community (December 2004), and no removal. Removal was done manually by divers. Before, immediately after first removal, and on eight occasions afterwards, divers counted corals, zoanthids and sponges in each plot, and removed invasive corals in the multiple removal treatment. Before removal, all plots had similar covers of sponge and zoanthids (apart from multiple-removal plots where zoanthids were absent).

(1) De Paula A.F., Fleury B.G., Lages B.G. & Creed J.C. (2017) Experimental evaluation of the effects of management of invasive corals on native communities. *Marine Ecology Progress Series,* 572, 141–154.

8.20. Use of non-native, invasive or other problematic species from populations established in the wild for recreational or commercial purposes

- We found no studies that evaluated the effects of using non-native, invasive or other problematic species from populations established in the wild for recreational or commercial purposes on subtidal benthic invertebrate 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 non-native, invasive or other problematic species have potential recreational or commercial use and therefore could be valuable. For such species, such as the edible Pacific oyster *Crassostrea* (also known as *Magallana*) *gigas* or Wakame kelp, *Undaria pinnatifida* in the UK (Epstein & Smale, 2017), management of populations established in the wild could potentially be through intentional recreational or commercial harvest (Nuñez *et al*. 2012; Pasko *et al*. 2014). For instance, a campaign called "Eat Lionfish" in 2010 aimed to promote the capture of the lionfish *Pterois volitans*, which is invasive in many parts of the world, for human consumption (Franke 2007; Nuñez *et al.* 2012). However, it is possible that enabling collection/hunting/fishing of introduced, non-native, or problematic marine species can lead to unintentional, perverse incentives to maintain an invasive population (Campbell *et al.* 2009). This may lead to acceptance of introduced marine species, with a reduced motivation to act to eradicate and/or manage invasive, non-native, and problematic species, which is often against International Treaties that a country may be signatory to (Campbell *et al.* 2009). Experts advise that a balance needs to be considered and struck between controlling/eradicating and creating unintentional perverse incentives to maintain a population of non-native, introduced, or problematic marine species (Simberloff *et al.* 2011).

Campbell M.L., Grage A., Mabin C. & Hewitt C.L. (2009) Conflict between International Treaties: Failing to mitigate the effects of introduced marine species. *Dialogue,* 28, 46–56

Epstein G. & Smale D.A. (2017) *Undaria pinnatifida*: A case study to highlight challenges in marine invasion ecology and management. *Ecology and Evolution*, 7, 8624–8642.

Franke J.M. (2007) *The invasive species cookbook: conservation through gastronomy.* Bradford street Press, Wauwatosa, WI.

Kilian J.V., Klauda R.J., Widman S., Kashiwagi M., Bourquin R., Weglein S. & Schuster J. (2012) An assessment of a bait industry and angler behavior as a vector of invasive species. *Biological Invasions,* 14, 1469–1481.

Nuñez M.A., Kuebbing S., Dimarco R.D. & Simberloff D. (2012) Invasive species: to eat or not to eat, that is the question. *Conservation Letters*, 5, 334–341.

Pasko S., Goldberg J., MacNeil C. & Campbell M. (2014) Review of harvest incentives to control invasive species. *Management of Biological Invasions*, 5, 263–277.

Simberloff D., Alexander J., Allendorf F., Aronson J., Antunes P.M., Bacher S., Bardgett R., Bertolino S., Bishop M., Blackburn T.M. & Blakeslee A. (2011) Non-natives: 141 scientists object. *Nature*, 475, 7354.

9. Threat: Pollution

Background

Pollution of the marine environment can originate from a multitude of sources and is generally agreed to have major direct and indirect negative impacts on the marine environment (Clark *et al.* 2001; Islam & Tanaka 2004), with consequences for subtidal benthic invertebrates (Rainbow 2017). Sources of pollution include domestic and urban wastewaters, industrial and military effluents, intensive aquaculture systems and run-offs from land agriculture, garbage and solid wastes, and pollution from excess energy such as light, noise and thermal pollution (Clark *et al.* 2001). This chapter describes the evidence for interventions designed to prevent, reduce, or mitigate the threat from various pollution sources.

Islam M.S. & Tanaka M. (2004) Impacts of pollution on coastal and marine ecosystems including coastal and marine fisheries and approach for management: a review and synthesis. *Marine Pollution Bulletin*, 48, 624–649.

Clark R.B., Frid C. & Attrill M. (2001) *Marine pollution* (5th ed). Oxford University Press, Oxford; New York.

Rainbow P.S. (2017) Heavy metal levels in marine invertebrates. Pages 67–79 in: *Heavy metals in the marine environment*. CRC press.

General

9.1. Transplant/translocate 'bioremediating' species

- We found no studies that evaluated the effects of transplanting and/or translocating bioremediating species on subtidal benthic invertebrate 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 sources of pollution, for instance from sewage outfalls, aquaculture farms, or agriculture wastes in watercourses, can cause an excess in nutrients leading to eutrophication, phytoplankton blooms, and reduction in water quality such as reduced light and oxygen. This type of pollution can be biologically 'remediated' (reverse, removed, or counteracted) by transplanting or translocating particular species to the affected area (Sode *et al.* 2013). These species, called '*bioremediating species*' can naturally improve water quality through feeding (for instance filter-feeding species such as mussels), or through photosynthesis (for instance algae species) (Chung *et al*. 2002). Transplanting or translocating such species to a polluted area may reduce pollution levels and allow subtidal benthic invertebrate communities to recover over time (Sode *et al.* 2013).

Evidence for interventions related to pollution bioremediation are summarised under "Threat: Pollution – Use other bioremediation methods in aquaculture", and evidence related to transplantation and/or translocation of species are summarised under the chapter "Species management".

Chung I.K., Kang Y.H., Yarish C., Kraemer G.P. & Lee J.A. (2002) Application of seaweed cultivation to the bioremediation of nutrient-rich effluent. *Algae,* 17, 187–194.
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.

9.2. Add chemicals or minerals to sediments to remove or neutralise pollutants

- **Two studies** examined the effects of adding chemicals or minerals to sediments to remove or neutralise pollutants on subtidal benthic invertebrate populations. Both studies evaluated the use of coal ash in Hiroshima Bay[1,2] (Japan).

COMMUNITY RESPONSE (1 STUDY)

- **Overall richness/diversity (1 study):** One controlled, before-and-after study in Hiroshima Bay[1] found that adding coal ash increased invertebrate species richness in winter but not summer compared to untreated sites.

POPULATION RESPONSE (2 STUDIES)

- **Overall abundance (2 studies):** One controlled, before-and-after study in Hiroshima Bay[1] found that adding coal ash increased invertebrate abundance in winter but not summer compared to untreated sites. One controlled study in Hiroshima Bay[2] found that one of two types of coal ash increased combined invertebrate and fish abundance, but not biomass.

Background

Marine sediments can accumulate pollutants over time, such as those leaching from aquaculture systems, sewage outfalls, or nearby agriculture fields, and negatively affect subtidal benthic invertebrates. Chemicals or minerals can be added to sediments to reduce or remove pollutants within the sediments (Shin & Kim 2016; Yamamoto *et al*. 2013). For example, granulated coal ash can be used with the aim of reducing concentrations of phosphates and hydrogen sulphide (Kim *et al.* 2014). This may reduce pollution levels in the sediment at the treated area and allow subtidal benthic invertebrate communities to recover over time (Kim *et al.* 2014).

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.
Yamamoto, T., Harada, K., Kim, K.H., Asaoka, S., & Yoshioka, I. (2013) Suppression of phosphate release from coastal sediments using granulated coal ash. *Estuarine, Coastal and Shelf Science*, 116, 41–49.

A controlled, before-and-after study in 2008–2010 in one area of soft seabed in Hiroshima Bay, Japan (1) found that adding coal ash to sediments to remove phosphate and hydrogen sulphide appeared to result in more species and individual invertebrates compared to before treatment and to adjacent untreated sites, during winter but not summer. However, results were not statistically tested. In winter, species richness increased (post-treatment: 17–22; pre-treatment: 8; untreated: 0–11/sample), and invertebrate abundance increased (post-treatment: 3,345–3,859; pre-treatment: 42; untreated: 0–507/m^2). In summer, species richness and invertebrate abundance were similar in post-treatment sites (species: 3–7/sample; abundance: 49–944/m^2), pre-treatment sites (species: 2 /sample; abundance: 204/m^2), and untreated sites (species: 0–6/sample; abundance: 0–261/m^2). Annually between August 2008 and August 2012 (except 2009), two sites were sampled once in winter and once in summer (one sample/site/time point). In May 2010, coal ash was scattered onto the sediment at one site to a depth of 10 cm; the other site was untreated. At the treated site, sediment samples were collected using a 25 x 25 cm quadrat to a depth of 10 cm. At the untreated site, sediment samples were collected using a sediment grab (dimensions unspecified). Invertebrates (> 1 mm) were identified and counted.

A controlled study in 2008–2011 in one area of soft seabed in Hiroshima Bay, Japan (2) found that adding coal ash to sediments to remove hydrogen sulphide increased combined invertebrate and fish abundance compared to untreated sediments in one of two comparisons, but did not change overall biomass over three years. Abundance at the site treated with Osaki coal ash was greater (41–496 individual/quadrat) than at the untreated site (14–281). The site treated with Onoda coal ash had similar abundance (29–262) to the untreated site. Combined invertebrate and fish biomass at the treated sites were similar (Osaki: 0.3–8.5 unit unspecified; Onoda: 0.3–9) to that of the untreated site (untreated: 0.6–13). In October 2008, two sites (75 x 50 m; 80 m apart) were treated with one of two types of coal ash (Onoda or Osaki; see study for details) to a depth of 20 cm and a third site (50 m away) was not treated. Every three months between February 2009 and November 2011, three sediment samples were collected at each site using a 25 x 25 cm quadrat to a depth of 20 cm. Both invertebrates and fish (>1 mm) were identified, counted, and weighed.

(1) 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.
(2) Yamamoto T., Kim K. H. & Shirono K. (2015). A pilot study on remediation of sediments enriched by oyster farming wastes using granulated coal ash. *Marine Pollution Bulletin*, 90, 54–59.

9.3. Establish pollution emergency plans

- We found no studies that evaluated the effects of establishing pollution emergency plans on subtidal benthic invertebrate 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

Sudden acute pollution events, for instance oil spills, can cause serious disturbances and harm to marine life (White *et al.* 2012). Pollution emergency plans provide an overview of possible procedures, as well as details of which authorities to contact, in case of a pollution event. The aim of emergency plans is to increase the speed and effectiveness of response, should a pollution event occur (Li *et al*. 2016; Qiao *et al*. 2002). Establishing emergency plans may benefit subtidal benthic invertebrates through a faster response and a more efficient control of the pollution, should such event occur.

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.
Qiao B., Chu J.C., Zhao P., Yu Y. & Li Y. (2002) Marine oil spill contingency planning. *Journal of Environmental Sciences*, 14, 102–107.

White H.K., Hsing P.Y., Cho W., Shank T.M., Cordes E.E., Quattrini A.M., Nelson R.K., Camilli R., Demopoulos A.W., German C.R. & Brooks J.M. (2012) Impact of the Deepwater Horizon oil spill on a deep-water coral community in the Gulf of Mexico. *Proceedings of the National Academy of Sciences*, 109, 20303–20308.

Domestic and urban wastewater

9.4. Limit, cease or prohibit the dumping of untreated sewage

• We found no studies that evaluated the effects of limiting, ceasing or prohibiting the dumping of untreated sewage on subtidal benthic invertebrate 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

Untreated sewage reaching the marine environment can impact subtidal benthic invertebrates through the introduction of bacteria, excess nutrients, toxic substances and solid particles, and through changes in salinity (McGann *et al.* 2003). Limiting, ceasing or prohibiting the dumping of untreated sewage in an area may benefit subtidal benthic invertebrates by reducing or stopping the source of pollution and allowing communities to potentially recover over time (Bustamante *et al.* 2012).

Evidence for other interventions related to sewage pollution are summarised under "Threat: Pollution – Limit, cease or prohibit the dumping of sewage sludge", "Set or improve minimum sewage treatment standards", and "Limit the amount of storm wastewater overflow".

Bustamante M., Bevilacqua S., Tajadura J., Terlizzi A. & Saiz-Salinas J.I. (2012) Detecting human mitigation intervention: Effects of sewage treatment upgrade on rocky macrofaunal assemblages. *Marine Environmental Research,* 80, 27–37.

McGann M., Alexander C.R. & Bay S.M. (2003) Response of benthic foraminifers to sewage discharge and remediation in Santa Monica Bay, California. *Marine Environmental Research*, 56, 299–342.

9.5. Limit, cease or prohibit the dumping of sewage sludge

• **Two studies** examined the effects of ceasing or prohibiting the dumping of sewage sludge on subtidal benthic invertebrate populations. One study was in the New York Bight[1] (USA), one in the North Sea[2] (UK).

COMMUNITY RESPONSE (2 STUDIES)

- **Overall community composition (2 studies):** One before-and-after, site comparison study in the New York Bight[1] found that after ceasing sewage sludge dumping, overall invertebrate community composition became more similar to less disturbed sites. One replicated, site comparison study in the North Sea[2] found that overall invertebrate community composition changed but remained different to that of natural sites.

POPULATION RESPONSE (2 STUDIES)

- **Overall abundance (1 study):** One replicated, site comparison study in the North Sea[2] found that after ceasing sewage sludge dumping, overall invertebrate abundance became similar to that of natural sites.

- **Worm abundance (1 study):** One before-and-after, site comparison study in the New York Bight[1] found that after ceasing sewage sludge dumping, abundance of pollution-indicator polychaete worms decreased and became similar to that of natural sites.

Background

Sewage sludge is the residual, semi-solid material produced as a by-product during sewage treatment. Sewage sludge can be disposed of at sea and can impact subtidal benthic invertebrates through the introduction of bacteria, heavy metals and chemicals (McGann *et al.*, 2003). Limiting, ceasing or prohibiting the dumping of sewage sludge in an area may potentially benefit subtidal benthic invertebrates by reducing or stopping the source of pollution and allowing communities to potentially recover over time (Birchenough & Frid 2009). Evidence for related interventions is summarised under "Threat: Pollution – Domestic and urban wastewater".

Birchenough S.N. & Frid C.L. (2009) Macrobenthic succession following the cessation of sewage sludge disposal. *Journal of Sea Research*, 62, 258–267.

McGann M., Alexander C.R. & Bay S.M. (2003) Response of benthic foraminifers to sewage discharge and remediation in Santa Monica Bay, California. *Marine Environmental Research*, 56, 299–342.

A before-and-after, site comparison study in 1987–1989 of three sandy sites in the inner New York Bight, North Atlantic Ocean, USA (1) found that over the 21 months after sewage-sludge dumping ceased, invertebrate community composition became more similar to that of historically less-disturbed sites. Community data were reported as graphical analyses and statistical model results. In addition, the abundance of the pollution-indicator polychaete worm *Capitella* spp. decreased after dumping had ceased (before: 0–3,000; after: 0–43 individuals/0.1 m^2) to similar levels as natural sites (approximately 0). Community composition at the less-disturbed sites remained stable over time. In 1987, dumping of sewage sludge in an area 22 km off the coast stopped after 63 years of activity. Monthly in July 1986–December 1987 (before complete cessation) and in January 1988–December 1989 (after cessation), one impacted site and two adjacent sites (low impact; no impact) were sampled at 29–31 m depths using a 0.1 m^2 sediment grab. Each time, three samples were collected, and invertebrates (>0.5 mm) identified and counted.

A replicated, site comparison study in 1999–2001 in one soft seabed area along the Northumberland coast, North Sea, UK (2) found that ceasing the disposal of sewage sludge led to changes in invertebrate community composition and decreases in overall invertebrate abundance over time. Community composition at the sewage sites changed over the three years after disposal stopped but remained different to that of the natural sites (data presented as graphical analyses and statistical model results). After one year, invertebrate abundance had decreased at the sewage sites (169–194 individuals/0.1 m^2) compared to three months after sewage dumping stopped (245–405), and was similar to that of natural sites (180–188). In December 1998, disposal of sewage sludge ceased at a site 10–13 km off the coast. Between 1999 and 2001 samples were collected annually in March, August, and December (except March 2000). Five samples were collected using sediment grabs (0.1 m^2) at each of four sites: two located at the sewage site, and two natural sites located 9–10 km away. Invertebrates (>0.5 mm) were identified and counted.

(1) Vitaliano J.J., Fromm S.A., Packer D.B., Reid R.N. & Pikanowski R.A. (2007) Recovery of benthic macrofauna from sewage sludge disposal in the New York Bight. *Marine Ecology Progress Series*, 342, 27–40.
(2) Birchenough S.N. & Frid C.L. (2009) Macrobenthic succession following the cessation of sewage sludge disposal. *Journal of Sea Research*, 62, 258–267.

9.6. Set or improve minimum sewage treatment standards

- **One study** examined the effects of improving minimum sewage treatment standards on subtidal benthic invertebrates. The study was in the Bay of Biscay[1] (Spain).

COMMUNITY RESPONSE (1 STUDY)

- **Overall community composition (1 study):** One before-and-after, site comparison study in the Bay of Biscay[1] found that after introducing a secondary treatment of sewage wastewaters, invertebrate community composition at an impacted site did not significantly change compared to unimpacted sites.
- **Overall richness/diversity (1 study):** One before-and-after, site comparison study in the Bay of Biscay[1] found that after introducing a secondary treatment of sewage wastewaters, invertebrate richness and diversity at an impacted site did not significantly change compared to unimpacted sites.

POPULATION RESPONSE (1 STUDY)

- **Overall abundance (1 study):** One before-and-after, site comparison study in the Bay of Biscay[1] found that after introducing a secondary treatment of sewage wastewaters, total cover of invertebrates significantly increased at an impacted site at 8 m but not 3 m depth, compared to unimpacted sites.

Background

Untreated sewage reaching the marine environment can impact subtidal benthic invertebrates through the introduction of bacteria, excess nutrients, toxic substances and solid particles, and through changes in salinity (McGann *et al.* 2003). Setting minimum sewage treatment standards, or improving the standards already in place, could potentially ensure that pollution level and associated risks to subtidal benthic invertebrates are minimized. For instance, improving standards can be achieved by installing a secondary treatment involving the mechanical and biological removal of settleable solids and dissolved organic compounds (Bustamante *et al.* 2012).

Evidence for related interventions is summarised under “Threat: Pollution – Limit, cease or prohibit the dumping of untreated sewage”, “Limit, cease or prohibit the dumping of sewage sludge” and “Limit the amount of storm wastewater overflow”.

Bustamante M., Bevilacqua S., Tajadura J., Terlizzi A. & Saiz-Salinas J.I. (2012) Detecting human mitigation intervention: Effects of sewage treatment upgrade on rocky macrofaunal assemblages. Marine Environmental Research, 80, 27-37
McGann M., Alexander C.R. & Bay S.M. (2003) Response of benthic foraminifers to sewage discharge and remediation in Santa Monica Bay, California. *Marine Environmental Research*, 56, 299–342.

A before-and-after, site comparison in 2001–2009 of four rocky seabed sites in Plentzia Bay, southern Bay of Biscay, northern Spain (1) found that improving the treatment of sewage wastewaters before discharge at one impacted site did not result in changes in invertebrate community composition or diversity after three years. Community composition did not change over time at the impacted site nor at three adjacent unimpacted sites, and communities appeared to be similar at all sites both before and after sewage treatment improvement (data reported as statistical model results and graphical analyses). In addition, diversity did not change at the impacted site or unimpacted sites over time (data reported as five diversity indices). Total species cover significantly increased at 8 m depth at the impacted site (before: 14–20%; after: 42–46%) compared to the unimpacted site (before: 3–42%; after: 4–42%), but not at 3 m depth where cover changed similarly at the impacted site (before: 11–20%; after: 31–63%) and the unimpacted sites (before: 5–50%; after: 23–98%). Raw sewage had been released into the intertidal area at the study area for 40 years until physical-chemical treatment was introduced in 1998. In 2006, a secondary biological treatment was introduced. Every two years between 2001 and 2009, one impacted site and three adjacent unimpacted sites were surveyed. Six locations/site were surveyed in summer (three at 3 m depth, three at 8 m). Invertebrate species were counted, and their cover visually estimated in three 40 x 40 cm quadrats.

(1) Bustamante M., Bevilacqua S., Tajadura J., Terlizzi A. & Saiz-Salinas J.I. (2012) Detecting human mitigation intervention: Effects of sewage treatment upgrade on rocky macrofaunal assemblages. *Marine Environmental Research*, 80, 27–37.

9.7. Limit the amount of storm wastewater overflow

- We found no studies that evaluated the effects of limiting the amount of storm wastewater overflow on subtidal benthic invertebrate 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 sewer systems collect rainwater runoff, sewage, and industrial wastewater in the same pipe, where it is then transported to a sewage treatment plant. During heavy rainfall events or snow melt the volume of wastewater can exceed the capacity of treatment facilities. In such instances, sewer systems can overflow and discharge untreated storm water and wastewater directly into rivers and seas (Moffa 1997). Untreated storm and wastewater can impact subtidal benthic invertebrates through the introduction of bacteria, excess nutrients, toxic substances, solid particles and changes in salinity (Bustamante *et al.* 2012). Limiting the amount of untreated storm and waste waters overflowing, for instance by increasing the capacity of treatment facilities, can potentially reduce pollution levels and associated risks to subtidal benthic invertebrates (Field & Struzeski 1972).

Evidence for interventions related to sewage pollution are summarised under "Threat: Pollution – Limit, cease or prohibit the dumping of untreated sewage", "Limit, cease or prohibit the dumping of sewage sludge" and "Set or improve minimum sewage treatment standards".

Bustamante M., Bevilacqua S., Tajadura J., Terlizzi A. & Saiz-Salinas J.I. (2012) Detecting human mitigation intervention: Effects of sewage treatment upgrade on rocky macrofaunal assemblages. *Marine Environmental Research,* 80, 27–37.
Field R. & Struzeski Jr. E. J. (1972) Management and control of combined sewer overflows. *Journal (Water Pollution Control Federation)*, 1393–1415.
Moffa P.E. (Ed.). (1997) *The control and treatment of combined sewer overflows*. John Wiley & Sons.

Industrial and military effluents

9.8. Use double hulls to prevent oil spills

• We found no studies that evaluated the effects of using double hulls to prevent oil spills on subtidal benthic invertebrate 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

Oil spills can be disastrous to marine life, including subtidal benthic invertebrates (White *et al.* 2012). Double hulls, where the bottom and sides of ships have two layers of watertight surfaces, can be used to prevent oil spills and have been required in some countries since the 1990s (Alcock 1992). Double hulls can reduce vessel damage to tankers when involved in accidents, and their use has been shown to significantly reduce the number of oil spills (Glen 2010; Yip *et al.* 2011). Using double hulls may potentially reduce the risks to subtidal benthic invertebrates from pollution following accidental oil spills.

Evidence for other interventions related to oil pollution are summarised under "Threat: Pollution – Remove or clean-up oil pollution following a spill".

Alcock T.M. (1992) Ecology tankers and the Oil Pollution Act of 1990: A history of efforts to require double hulls on oil tankers. *Ecology Law Quaterly.* 19, 97.

Glen D. (2010) Modelling the impact of double hull technology on oil spill numbers. *Maritime Policy & Management*, 37, 475–487.

Yip T.L., Talley W.K. & Jin D. (2011) The effectiveness of double hulls in reducing vessel-accident oil spillage. *Marine Pollution Bulletin,* 62, 2427–2432.

White H.K., Hsing P.Y., Cho W., Shank T.M., Cordes E.E., Quattrini A.M., Nelson R.K., Camilli R., Demopoulos A.W., German C.R. & Brooks J.M. (2012) Impact of the Deepwater Horizon oil spill on a deep-water coral community in the Gulf of Mexico. *Proceedings of the National Academy of Sciences*, 109, 20303–20308.

9.9. Remove or clean-up oil pollution following a spill

- **One study** examined the effects of removing and cleaning-up oil pollution following a spill on subtidal benthic invertebrates. The study was in the Baltic Proper[1] (Sweden).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Mollusc condition (1 study):** One replicated, controlled, before-and-after study in the Baltic Proper[1] found that after cleaning-up spilled oil using high pressure hot water, crude oil content increased in mussels and did not naturally decrease over time, and was higher than in mussels from an uncleaned contaminated and a non-contaminated site.

Background

Oil spills can be disastrous to marine life, including subtidal benthic invertebrates (White *et al*. 2012). The control and remediation of oil spills can be undertaken in a multitude of ways: for instance by using booms (floating barriers that contain a spill to a delimited zone) and skimmers (devices that collect and remove oil) to remove oil pollution from the surface of the water, using dispersants that break oil into small droplets (Hartwick *et al*. 1982), using sorbents, or using controlled burning of the oil (Al-Majed *et al*. 2012). Different methods have different outcomes and side-effects, but when successful may potentially reduce the risks of toxicity and direct harm to subtidal benthic invertebrates.

Evidence for related interventions is summarised under "Threat: Pollution – Use double hulls to prevent oil spills" and "Establish pollution emergency plans".

Al-Majed A.A., Adebayo A.R. & Hossain M.E. (2012) A sustainable approach to controlling oil spills. *Journal of Environmental Management*, 113, 213–227.

Hartwick E.B., Wu R.S.S. & Parker D.B. (1982) Effects of a crude oil and an oil dispersant (Corexit 9527) on populations of the littleneck clam (*Protothaca staminea*). *Marine Environmental Research*, 6, 291–306.

White H.K., Hsing P.Y., Cho W., Shank T.M., Cordes E.E., Quattrini A.M., Nelson R.K., Camilli R., Demopoulos A.W., German C.R. & Brooks J.M. (2012) Impact of the Deepwater Horizon oil spill on a deep-water coral community in the Gulf of Mexico. *Proceedings of the National Academy of Sciences*, 109, 20303–20308.

A replicated, controlled, before-and-after study in 1983 in one area of rocky coastline in the northern Baltic Proper, Sweden (1) found that high pressure hot water shore cleaning technique following an oil spill tended to increase crude oil content of blue mussels *Mytilus edulis*. Results were not statistically tested. After three days, petroleum hydrocarbon content (crude oil) appeared to have increased in mussels from 40 μg/g to 533–657 μg/g, and decreased by only approximately 20–45% (to 290–530 μg/g) after two weeks. These levels tended to be higher than in mussels from an adjacent uncleaned contaminated site (43 μg/g) and mussels from a non-contaminated site (30 μg/g). In summer 1980, crude oil was experimentally spilled on the shore and cleaned. The "cleaned" sea area directly off the shore was fenced with booms, and sorption agents used on the sea surface. Blue mussels (>30 mm in length) collected from a non-contaminated site were placed in 11 net bags (12/bag). A week before cleaning, nine bags were placed within the fenced area, one bag at an uncleaned contaminated site, and one bag at the non-contaminated site, all at 0.5 m depth. One fenced bag was retrieved before cleaning. Three days after cleaning, the bag from the uncleaned contaminated site was retrieved, as well as six bags from the cleaned area. After two weeks, all remaining bags were retrieved. The crude oil content of mussels was measured.

(1) Ganning B., Broman D. & Lindblad C. (1983) Uptake of petroleum hydrocarbons by the blue mussel (*Mytilus edulis* L.) after experimental oiling and high pressure, hot water shore cleaning. *Marine Environmental Research*, 10, 245–254.

9.10. Set regulatory ban on marine burial of nuclear waste

- We found no studies that evaluated the effects of setting regulatory ban on marine burial of nuclear waste on subtidal benthic invertebrate 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

The disposal of nuclear and radioactive waste at sea was practised by 13 countries from 1946 to 1993, until it was banned in 1993 following international treaties. However, enforcement is lacking in parts of the world, where illegal dumping is reported to occur.

Disposal within the sediment (sub-sea burial) was never implemented and such process currently falls under the ban of nuclear waste disposal at sea. However, it is being proposed by various countries and may be an option in the future (Hollister & Nadis 1998). Setting pre-emptive regulatory bans on the sub-sea burial of nuclear waste can help prevent the occurrence of associated threats to subtidal benthic invertebrates.

Hollister C.D. & Nadis S. (1998) Burial of radioactive waste under the seabed. *Scientific American*, 278, 60–65.

Aquaculture effluents

9.11. Cease or prohibit aquaculture activity

- **Two studies** examined the effects of ceasing or prohibiting aquaculture activity on subtidal benthic invertebrate populations. Both studies were in the Mediterranean Sea[1,2] (Italy and Spain).

COMMUNITY RESPONSE (2 STUDIES)

- **Overall community composition (1 study):** One before-and-after, site comparison study in the Mediterranean Sea[1] found that after ceasing aquaculture activity invertebrate community composition remained different to that of an unfarmed site.
- **Worm community composition (1 study):** One before-and-after, site comparison study in the Mediterranean Sea[2] found that after ceasing aquaculture activity worm community composition community composition remained different to that of an unfarmed site.

POPULATION RESPONSE (2 STUDIES)

- **Overall abundance (1 study):** One before-and-after, site comparison study in the Mediterranean Sea[1] found that after ceasing aquaculture activity overall invertebrate abundance was similar to an unfarmed site.
- **Worm abundance (1 study):** One before-and-after, site comparison study in the Mediterranean Sea[2] found that after ceasing aquaculture activity abundance of health-indicating worms increased, and abundance of pollution-indicating worms decreased.

Background

Aquaculture systems can negatively impact subtidal benthic invertebrate communities through pollution and diminished water quality (Wu *et al*. 1994). Ceasing or prohibiting aquaculture activity in an area, for instance by decommissioning farms or relocation to a different area, would remove the source of pollution and potentially allow for subtidal benthic invertebrate communities to recover over time (Johannessen *et al.* 1994). Aquaculture systems also pose serious environment risks by promoting the spread of non-native, invasive, and pest species and diseases.

Evidence for interventions related to non-native, invasive and pest species is summarised in "Threat: Invasive and other problematic species, genes and diseases – Aquaculture".

Johannessen P., Botnen H. & Tvedten Ø.F. (1994) Macrobenthos: before, during and after a fish farm. *Aquaculture Research*, 25, 55–66

Wu R.S.S., Lam K.S., MacKay D.W., Lau T.C. & Yam V. (1994) Impact of marine fish farming on water quality and bottom sediment: A case study in the sub-tropical environment. *Marine Environmental Research*, 38, 115–145.

A before-and-after, site comparison study in 1997 of two soft seabed sites in the Gulf of Gaeta, Mediterranean Sea, Italy (1) found that after removing a fish farm, invertebrate abundance appeared similar to that of an unfarmed site after two months, but community composition remained different after four months. Before removal, abundance at the farmed site (850–1,350/10 cm^2) appeared different to the unfarmed site (1,250–2,750). This was still true a month after removal (farmed: 1,350; unfarmed: 2,800). After two months, abundances were similar at all sites (farmed: 1,500–2,300; unfarmed: 2,000–2,850). Community composition remained different after four months (data presented as graphical analyses). A fish farm was removed in July 1997. One farmed site and one unfarmed site (1 km north) were surveyed monthly between March and October 1997. Three sediment samples were taken by divers at each site during each survey using a core (3.7 cm diameter, 10 cm depth). Invertebrates (between 37 μm and 1 mm) were identified and counted.

A before-and-after, site comparison study in 2007–2008 in three soft seabed locations 4.8 km off the coast of Murcia, Mediterranean Sea, southeast Spain (2) found that eight months after removing a fish farm, the worm community had changed but was still different from that of two nearby unfarmed sites. The similarity in worm community between the farmed and unfarmed sites did not increase after removal (before: 43%; after: 41% similarity). However, abundance of opportunistic (indicating pollution) Capitellidae species decreased, while abundances of Onuphidae and Sabellidae species (indicating good health of sediment) increased at the farmed site after eight months (abundances not reported). A fish farm was dismantled in November 2007. One farmed site and two unfarmed sites (1 km and 1.3 km from the farmed site) were surveyed twice before (January and July 2007) and twice after (January and July 2008) dismantling. Four sediment samples were taken by divers at each site during each survey using a hand grab (20 x 10 x 10 cm). Worms (> 0.5 mm) were identified to family level and counted.

(1) Mazzola A., Mirto S., La Rosa T., Fabiano M. & Danovaro R. (2000) Fish-farming effects on benthic community structure in coastal sediments: analysis of meiofaunal recovery. *ICES Journal of Marine Science*, 57, 1454–1461.
(2) Aguado-Giménez F., Piedecausa M.A., Gutiérrez J.M., García-Charton J.A., Belmonte A. & García-García B. (2012) Benthic recovery after fish farming cessation: a "beyond-BACI" approach. *Marine Pollution Bulletin*, 64, 729–738.

9.12. Reduce aquaculture stocking densities

- We found no studies that evaluated the effects of reducing aquaculture stocking densities on subtidal benthic invertebrate 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

Aquaculture systems can negatively impact subtidal benthic invertebrate communities through pollution and diminished water quality (Wu *et al.* 1994). To limit the amount of pollution emitted by one aquaculture site, stocking density could be reduced (Naylor *et al*. 2000). A lower number of organisms in one area will produce less waste and limit organic enrichment. This may potentially reduce the level of impact in the vicinity of the aquaculture site and allow some recovery of the subtidal benthic invertebrate community.

Naylor R.L., Goldburg R.J., Primavera J.H., Kautsky N., Beveridge M.C., Clay J., Folke C., Lubchenco J., Mooney H. & Troell M. (2000) Effect of aquaculture on world fish supplies. *Nature*, 405, 1017.
Wu R.S.S., Lam K.S., MacKay D.W., Lau T.C. & Yam V. (1994) Impact of marine fish farming on water quality and bottom sediment: A case study in the sub-tropical environment. *Marine Environmental Research,* 38, 115–145.

9.13. Locate aquaculture systems in already impacted areas

- We found no studies that evaluated the effects of locating aquaculture systems in already impacted areas on subtidal benthic invertebrate 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

Aquaculture systems can negatively impact subtidal benthic invertebrate communities through pollution and diminished water quality (Wu *et al.* 1994). By locating aquaculture systems in areas that already have poor water quality (for instance due to sewage outfall), the source of pollution is restricted to that already impacted zone. This may potentially relieve other areas from additional pollution, without significantly further degrading the already impacted area which likely already holds an impacted subtidal benthic invertebrate community.

Evidence for other interventions related to the relocation of aquaculture activities are summarised under "Threat: Pollution – Locate aquaculture systems in locations with fast currents", "Locate aquaculture systems in vegetated areas", and "Locate artificial reefs near aquaculture systems (and vice versa) to act as biofilters".

Wu R.S.S., Lam K.S., MacKay D.W., Lau T.C. & Yam V. (1994) Impact of marine fish farming on water quality and bottom sediment: A case study in the sub-tropical environment. *Marine Environmental Research,* 38, 115–145.

9.14. Locate aquaculture systems in areas with fast currents

- We found no studies that evaluated the effects of locating aquaculture systems in areas with fast currents on subtidal benthic invertebrate 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

Aquaculture systems can negatively impact subtidal benthic invertebrate communities through pollution and diminished water quality (Wu *et al.* 1994). For instance, it can cause anoxic conditions (lack of oxygen) due to waste build up from fish food and faeces. Locating aquaculture systems in areas with fast currents may help maintain water quality by increasing water exchange, allowing greater dispersal and dilution of pollutant loads

(Hall-Spencer *et al.* 2006; Sarà *et al*. 2006). Reducing the risk of a built-up of pollution levels may prevent negative impacts on subtidal benthic invertebrates.

Evidence for other interventions related to the relocation of aquaculture activities are summarised under "Threat: Pollution – Locate aquaculture systems in already impacted areas", "Locate aquaculture systems in vegetated areas", and "Locate artificial reefs near aquaculture systems (and vice versa) to act as biofilters".

Hall-Spencer J., White N., Gillespie E., Gillham K. & Foggo A. (2006) Impact of fish farms on maerl beds in strongly tidal areas. *Marine Ecology Progress Series*, 326, 1–9.

Sarà G., Scilipoti D., Milazzo M. & Modica A. (2006) Use of stable isotopes to investigate dispersal of waste from fish farms as a function of hydrodynamics. *Marine Ecology Progress Series*, 313, 261–270.

Wu R.S.S., Lam K.S., MacKay D.W., Lau T.C. & Yam V. (1994) Impact of marine fish farming on water quality and bottom sediment: A case study in the sub-tropical environment. *Marine Environmental Research,* 38, 115–145.

9.15. Locate aquaculture systems in vegetated areas

- We found no studies that evaluated the effects of locating aquaculture systems in vegetated locations on subtidal benthic invertebrate 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

Aquaculture systems can negatively impact invertebrate subtidal communities through pollution and diminished water quality (Wu *et al.* 1994). Aquaculture systems can be located in areas with submerged vegetation, such as seagrass or kelp, which can help absorb the waste product effluents and mitigate the pollution originating from the installations (Mirto *et al*. 2010). This may help locally reduce or mitigate the deterioration in water quality in the area and benefit subtidal benthic invertebrates.

Evidence for other interventions related to the relocation of aquaculture activities are summarised under "Threat: Pollution – Locate aquaculture systems in already impacted areas", "Locate aquaculture systems in locations with fast currents", and "Locate artificial reefs near aquaculture systems (and vice versa) to act as biofilters".

Mirto S., Bianchelli S., Gambi C., Krzelj M., Pusceddu A., Scopa M., Holmer M. & Danovaro R. (2010) Fish-farm impact on metazoan meiofauna in the Mediterranean Sea: analysis of regional vs. habitat effects. *Marine Environmental Research*, 69, 38–47.

Wu R.S.S., Lam K.S., MacKay D.W., Lau T.C. & Yam V. (1994) Impact of marine fish farming on water quality and bottom sediment: A case study in the sub-tropical environment. *Marine Environmental Research,* 38, 115–145.

9.16. Moor aquaculture cages so they move in response to changing current direction

- We found no studies that evaluated the effects of mooring aquaculture cages so they move in response to changing current direction on subtidal benthic invertebrate 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

Aquaculture systems can negatively impact invertebrate subtidal communities through pollution and diminished water quality (Wu *et al.* 1994). Instead of mooring a cage in a fixed position, cages can be moored so they move in response to changes in currents. This may help disperse and dilute the accumulation of wastes and organic matter from fish food and faeces, thereby reducing the pollution levels in the area (Goudey *et al.* 2001; Sarà *et al*. 2006).

Goudey C.A., Loverich G., Kite-Powell H. & Costa-Pierce B.A. (2001) Mitigating the environmental effects of mariculture through single-point moorings (SPMs) and drifting cages. *ICES Journal of Marine Science,* 58, 497–503.

Sarà G., Scilipoti D., Milazzo M. & Modica A. (2006) Use of stable isotopes to investigate dispersal of waste from fish farms as a function of hydrodynamics. *Marine Ecology Progress Series*, 313, 261–270.

Wu R.S.S., Lam K.S., MacKay D.W., Lau T.C. & Yam V. (1994) Impact of marine fish farming on water quality and bottom sediment: A case study in the sub-tropical environment. *Marine Environmental Research,* 38, 115–145.

9.17. Leave a fallow period during fish/shellfish farming

- **Three studies** examined the effects of leaving a fallow period during fish farming on subtidal benthic invertebrate populations. Two studies were in the Tasman Sea[1,2] (Australia), and one in the North Pacific Ocean[3] (USA).

COMMUNITY RESPONSE (3 STUDIES)

- **Overall community composition (2 study):** Two replicated, before-and-after, site comparison study in the Tasman Sea[1,2] found that after a fallow period invertebrate community composition became similar to that occurring before the fish were added but remained different to communities at sites without fish farms.
- **Worm community composition (1 study):** One replicated, before-and-after, site comparison study in the North Pacific Ocean[3] found that after a fallow period polychaete worm community composition changed but remained different to communities at sites without fish farms.
- **Worm richness/diversity (1 study):** One replicated, before-and-after, site comparison study in the North Pacific Ocean[3] found that after a fallow period polychaete worm diversity did not change and remained lower compared to sites without fish farms.

POPULATION RESPONSE (2 STUDIES)

- **Worm abundance (2 studies):** Two replicated, before-and-after, site comparison studies in the Tasman Sea[2] and the North Pacific Ocean[3] found that following a fallow period, abundances of pollution-indicator polychaete worms decreased, but remained higher compared to sites without fish farms.

Background

Aquaculture systems can negatively impact invertebrate subtidal communities through pollution and diminished water quality (Wu *et al.* 1994). Fallow periods (temporary cessation of production) are often used in aquaculture to mitigate the environmental effects of pollution from organic enrichment due to the stocking of fish, i.e. wastes from fish food and faeces. By temporarily stopping production, pollution is reduced, potentially allowing invertebrate subtidal communities to naturally recover over time until production is resumed (Lin & Bailey-Brock 2008; Zhulay *et al.* 2015).

Other evidence related to aquaculture activities are summarised under “Threat: Pollution – Aquaculture” including “Cease or prohibit aquaculture activity”.

Lin D.T. & Bailey-Brock J.H. (2008) Partial recovery of infaunal communities during a fallow period at an open-ocean aquaculture. *Marine Ecology Progress Series,* 371, 65–72.

Wu R.S.S., Lam K.S., MacKay D.W., Lau T.C. & Yam V. (1994) Impact of marine fish farming on water quality and bottom sediment: A case study in the sub-tropical environment. *Marine Environmental Research,* 38, 115–145.

Zhulay I., Reiss K. & Reiss H. (2015) Effects of aquaculture fallowing on the recovery of macrofauna communities. *Marine Pollution Bulletin,* 97, 381–390.

A replicated, before-and-after, site comparison study in 2003–2004 at two soft seabed locations in the Tasman Sea, southeastern Tasmania, Australia (1 – same experimental set-up as 2) found that after a three-month fallow period, invertebrate community composition had changed at farmed sites. After the fallow period (no fish in cages), communities were different to that of the pre-fallow period (fish in cages), and similar to communities present before fish were added (empty cages). Community data were reported as statistical model results and graphical analyses. In addition, although similarity in invertebrate community between farmed sites and sites without fish farms (natural seabed) increased after fallow (from 25% to 31% similarity at one location, and from 11% to 27% at the other location), communities remained different. Sediment samples were collected using a grab (0.07 m^2). At each of the two locations, five samples were collected at farmed and unfarmed sites before fish were added, following nine months of fish farming (pre-fallow period), and following the three-month fallow period. Invertebrates (>1 mm) were identified and counted. This was repeated over a second farming/fallowing cycle.

A replicated, before-and-after, site comparison study in 2001–2003 at two soft seabed locations in the Tasman Sea, southeastern Tasmania, Australia (2 – same experimental set-up at 1) found that after a three-month fallow period invertebrate community composition had changed at farmed sites. After the fallow period (no fish in cages), communities were different to that of the pre-fallow period (fish in cages), and similar to communities present before fish were added (empty cages), but not to that of nearby sites without fish farms (natural seabed). Community data were reported as statistical model results and graphical analyses). Although not tested for statistical significance, at one location, abundances of three pollution-indicator polychaete worms tended to be lower after the fallow period (*Capitella capitata* pre-fallow: 17,248 post-fallow: 2,621; *Neanthes cricognatha* pre-fallow: 199 post-fallow: 94; *Maldanidae* sp. pre-fallow: 54 post-fallow: 0 individuals/m^2), but remained higher than at sites without fish farms (*Capitella capitata* 5; *Neanthes cricognatha* 4; *Maldanidae sp*. 0 individual/m^2). At the second location, abundances of the opportunistic worms *Capitella capitata* and *Nebalia longicornis* tended to be lower following the fallow period (*Capitella capitata* pre-fallow: 7,470 post-fallow: 5,525; *Nebalia longicornis:* pre-fallow: 14,902 post-fallow: 1,791 individuals/m^2) but remained higher than at sites without fish farms (*Capitella capitata:* 19; *Nebalia longicornis:* 0). Sediment samples were collected using a grab (0.07 m^2) at 20 m depth. At each of the two locations, five samples were collected at farmed and unfarmed (located 150 m away) sites before fish were added, following nine months of fish farming (pre-fallow period), and following the three-month fallow period. Invertebrates (>1 mm) were identified and counted.

A replicated, before-and-after, site comparison study in 2001–2007 in four sandy seabed locations off the coast of Hawai'i, North Pacific Ocean, USA (3) found that after a six-month fallow period polychaete worm diversity, abundances and community composition changed at farmed sites, but remained different from that of sites without fish farms. Community data were reported as statistical model results and graphical analyses. The cumulative relative abundance of three pollution-indicator worms, *Capitella capitata*, *Neanthes arenaceodentata*, and *Ophryotrocha adherens*, tended to be lower after the fallow period (5%), compared to before (70%), but remained higher than at sites without fish farms (0%) (results not statistically tested). Worm species diversity at farmed sites was not different at the end compared to the start of the fallow period, and remained lower than at sites without fish farms (data reported as a diversity index). Four aquaculture locations were surveyed, each with four farmed sites and two unfarmed. Sediment samples were collected 16 times between November 2001 and August 2006 (before the fallow period), twice during the fallow period (between August 2006 to March 2007), and once in May 2007 (after fish were restocked). Divers collected three to five sediment samples/collection/site using hand tube corers (11 cm diameter, to 5 cm depth) at 39–45 m depths. Polychaete worms (>0.5 mm) were identified and counted.

(1) Macleod C.K., Moltschaniwskyj N.A. & Crawford C.M. (2006) Evaluation of short-term fallowing as a strategy for the management of recurring organic enrichment under salmon cages. *Marine Pollution Bulletin*, 52, 1458–1466.
(2) Macleod C.K., Moltschaniwskyj N.A., Crawford C.M. & Forbes S.E. (2007) Biological recovery from organic enrichment: some systems cope better than others. *Marine Ecology Progress Series*, 342, 41–53.
(3) Lin D.T. & Bailey-Brock J.H. (2008) Partial recovery of infaunal communities during a fallow period at an open-ocean aquaculture. *Marine Ecology Progress Series*, 371, 65–72.

9.18. Improve fish food and pellets to reduce aquaculture waste production

- We found no studies that evaluated the effects of improving fish food and pellets to reduce aquaculture waste production on subtidal benthic invertebrate 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

Aquaculture systems can negatively impact invertebrate subtidal communities through pollution and diminished water quality (Wu *et al.* 1994). Fish food in aquaculture is an important source of pollution because it is not always consumed by the farmed species and may sink to the seabed, leading to localised organic enrichment. Improving fish food and pellets to reduce aquaculture waste may reduce localised pollution and the associated threats to subtidal benthic invertebrates. This could be achieved by improving pellet aggregate strength, meaning that the pellet is less likely to break up, or by reducing the sinking rate of feed and pellets, both allowing more time for the cultured species to consume the food, thereby reducing the amount reaching the seabed (Cho & Bureau 1997; Wu 1995).

Cho C.Y. & Bureau D.P. (1997) Reduction of waste output from salmonid aquaculture through feeds and feeding. *The Progressive Fish-Culturist,* 59, 155–160.

Wu R.S.S. (1995) The environmental impact of marine fish culture: Towards a sustainable future. *Marine Pollution Bulletin,* 31, 159–166.
Wu R.S.S., Lam K.S., MacKay D.W., Lau T.C. & Yam V. (1994) Impact of marine fish farming on water quality and bottom sediment: A case study in the sub-tropical environment. *Marine Environmental Research,* 38, 115–145.

9.19. Reduce the amount of pesticides used in aquaculture systems

- We found no studies that evaluated the effects of reducing the amount of pesticides used in aquaculture systems on subtidal benthic invertebrate 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

Pesticides are used in aquaculture to reduce or eliminate pests. For example, the carbaryl pesticide Sevin is commonly used in the USA to control ghost shrimp *Callianassa californiensis* and mud shrimp *Upogebia pugettensis* in oyster culture (Weston 2000). Pesticides, however, have the potential to negatively impact non-target species, such as subtidal benthic invertebrates. This has been shown in the salmon aquaculture, where pesticides used against sea lice caused harm to crustaceans and worms (Mayor *et al.* 2009; Waddy *et al*. 2002). The risks associated with the use of pesticides can be reduced by applying them smaller doses, less frequently or across a smaller area. This may reduce the negative effects on subtidal benthic invertebrates.

Evidence for interventions related to the use of antibiotics are summarised under "Threat: Pollution – Reduce the amount of antibiotics used in aquaculture systems".

Mayor D. J., Solan M., Martinez I., Murray L., McMillan H., Paton G. I. & Killham K. (2008). Acute toxicity of some treatments commonly used by the salmonid aquaculture industry to Corophium volutator and Hediste diversicolor: Whole sediment bioassay tests. *Aquaculture*, 285, 102–108.
Waddy S. L., Burridge L.E., Hamilton M.N., Mercer S.M., Aiken D.E. & Haya K. (2002) Emamectin benzoate induces molting in American lobster, *Homarus americanus*. *Canadian Journal of Fisheries and Aquatic Sciences*, *59*, 1096–1099.
Weston D.P. (2000). Ecological effects of the use of chemicals in aquaculture. Pages 23–30, in: J.R. Arthur, C.R. Lavilla-Pitogo, & R.P. Subasinghe (Eds.) *Use of Chemicals in Aquaculture in Asia: Proceedings of the Meeting on the Use of Chemicals in Aquaculture in Asia.* 20-22 May 1996. Tigbauan, Iloilo, Philippines: Aquaculture Department, Southeast Asian Fisheries Development Center.

9.20. Reduce the amount of antibiotics used in aquaculture systems

- We found no studies that evaluated the effects of reducing the amount of antibiotics used in aquaculture systems on subtidal benthic invertebrate 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

Antibiotics are used in aquaculture to reduce or eliminate harmful bacteria (Burridge *et al*. 2010). Because they are selected for specific species and bacteria, they usually have low toxicity to other organisms. However, some antibiotics have been shown to accumulate and persist in sediments, with potential negative effects to subtidal benthic invertebrates, such as antibiotic resistance (Burridge *et al*. 2010; Cabello 2006). The risks associated with the use of antibiotics can be reduced by applying them smaller doses, less frequently or across a smaller area. In addition, research has shown that alternatives to antibiotics can be used successfully in aquaculture (Defoirdt *et al.* 2011). This may reduce the negative effects on subtidal benthic invertebrates, or even remove the source of the threat, and allow for natural recovery.

Evidence for interventions related to the use of pesticides are summarised under "Threat: Pollution – Reduce the amount of pesticides used in aquaculture systems".

Burridge L., Weis J.S., Cabello F., Pizarro J. & Bostick K. (2010) Chemical use in salmon aquaculture: a review of current practices and possible environmental effects. *Aquaculture*, 306, 7–23.

Cabello F.C. (2006) Heavy use of prophylactic antibiotics in aquaculture: a growing problem for human and animal health and for the environment. *Environmental Microbiology,* 8, 1137–1144

Defoirdt T., Sorgeloos P. & Bossier P. (2011) Alternatives to antibiotics for the control of bacterial disease in aquaculture. *Current opinion in microbiology*, 14, 251–258.

9.21. Use species from more than one level of a food web in aquaculture systems

- We found no studies that evaluated the effects of using species from more than one level of a food web in aquaculture systems on subtidal benthic invertebrate 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

Integrated multi-trophic aquaculture is a type of aquaculture set-up where a number of complementary species from different levels of the food web are cultured at one site in order to optimize nutrient utilization and reduce waste ('Chávez-Crooker *et al*. 2010). It is considered an effective biological method of removing organic enrichment from aquaculture (bioremediation'; Chávez-Crooker *et al*. 2010; Naylor *et al.* 2000; Troell *et al*. 2009). In integrated multi-trophic aquaculture, the waste from one species becomes a source of energy for another. For instance, the waste of a fed finfish becomes the food of a filter-feeding mussel, whose waste is then taken up by sea urchins, while seaweeds also use the excess nutrients in the water through photosynthesis (Nobre *et al*. 2010). By moving toward a better ecosystem balance, such multi-trophic systems have the potential to improve water quality at aquaculture sites and benefit subtidal benthic invertebrates.

Chávez-Crooker P. & Obreque-Contreras J. (2010) Bioremediation of aquaculture wastes. *Current opinion in Biotechnology*, 21, 313–317.

Naylor R.L., Goldburg R.J., Primavera J.H., Kautsky N., Beveridge M.C., Clay J., Folke C., Lubchenco J., Mooney H. & Troell M. (2000) Effect of aquaculture on world fish supplies. *Nature*, 405, 1017.

Nobre A.M., Robertson-Andersson D., Neori A. & Sankar K. (2010) Ecological–economic assessment of aquaculture options: comparison between abalone monoculture and integrated multi-trophic aquaculture of abalone and seaweeds. *Aquaculture*, 306, 116–126.

Troell M., Joyce A., Chopin T., Neori A., Buschmann A.H. & Fang J.G. (2009). Ecological engineering in aquaculture—potential for integrated multi-trophic aquaculture (IMTA) in marine offshore systems. *Aquaculture*, 297, 1–9.

9.22. Locate artificial reefs near aquaculture systems (and vice versa) to act as biofilters

- We found no studies that evaluated the effects of locating artificial reefs near aquaculture systems to act as biofilters on subtidal benthic invertebrate 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

Aquaculture systems can negatively impact invertebrate subtidal communities through pollution and diminished water quality (Wu *et al.* 1994). Artificial reefs host a high biodiversity, including filter-feeding organisms, and stimulate the biological productivity around them, even in the surrounding soft sediments (Ambrose & Anderson 1990). By placing artificial reefs near aquaculture systems, they can act as natural biofilters following colonisation, improving water quality, and potentially benefitting surrounding subtidal benthic invertebrates (Aguado-Giménez *et al.* 2011; Angel *et al*. 2002; Gao *et al*. 2008).

Evidence of other interventions related to the relocation of aquaculture activities are summarised under "Threat: Pollution – Locate aquaculture systems in already impacted areas", "Locate aquaculture systems in locations with fast currents", and "Locate aquaculture systems in vegetated areas". Other evidence related to the creation of artificial reefs are summarised in the "Habitat restoration and creation" chapter.

Aguado-Giménez F., Piedecausa M.A., Carrasco C., Gutiérrez J.M., Aliaga V. & García-García B. (2011) Do benthic biofilters contribute to sustainability and restoration of the benthic environment impacted by offshore cage finfish aquaculture? *Marine Pollution Bulletin*, 62, 1714–1724.

Ambrose R.F. & Anderson T.W. (1990). Influence of an artificial reef on the surrounding infaunal community. *Marine Biology,* 107, 41–52.

Angel D.L., Eden N., Breitstein S., Yurman A., Katz T. & Spanier E. (2002) In situ biofiltration: a means to limit the dispersal of effluents from marine finfish cage aquaculture. *Hydrobiologia*, 469, 1–10.

Gao Q.-F., Shin P.K.S., Xu W.Z. & Cheung S.G. (2008). Amelioration of marine farming impact on the benthic environment using artificial reefs as biofilters. *Marine Pollution Bulletin*. 57, 652–661.

Wu R.S.S., Lam K.S., MacKay D.W., Lau T.C. & Yam V. (1994) Impact of marine fish farming on water quality and bottom sediment: A case study in the sub-tropical environment. *Marine Environmental Research,* 38, 115–145.

9.23. Use other bioremediation methods in aquaculture

- We found no studies that evaluated the effects of using other bioremediation methods in aquaculture on subtidal benthic invertebrate 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

Aquaculture systems can negatively impact invertebrate subtidal communities through pollution and diminished water quality (Wu *et al.* 1994). Various biological methods can be used to remove such pollution ('bioremediation'; Chávez-Crooker *et al*. 2010), including better managing the use and dosage of chemicals, better managing fish food, using integrated multi-trophic aquaculture, locating or relocating aquaculture farms to specific areas, or placing artificial reefs nearby to act as biofilters [these interventions are summarised above]. Other methods can be used to mitigate pollution from aquaculture, such as microbial nitrification and denitrification in sediments, or the use of technological applications such as mechanical and biological filters (Chávez-Crooker *et al*. 2010). Using bioremediation methods may help improve water quality at aquaculture sites and reduce the risks to subtidal benthic invertebrates from pollution.

Chávez-Crooker P. & Obreque-Contreras J. (2010) Bioremediation of aquaculture wastes. *Current Opinion in Biotechnology*, 21, 313–317.

Agricultural and forestry effluents

9.24. Regulate the use, dosage and disposal of agrichemicals

- We found no studies that evaluated the effects of regulating the use, dosage and disposal of agrichemicals on subtidal benthic invertebrate 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

Agrichemicals (or agrochemicals) are chemicals used in agriculture, such as pesticides, fertilisers and manure. They are designed to have long-lasting effects on living organisms, are often toxic to non-target species, and as such are considered a major source of pollution and toxicity in aquatic environments (Islam & Tanaka 2004; Rawlings *et al.* 1998). Agrichemicals can reach the marine environment through soil erosion, rivers, and other watercourse runoffs. There, they can accumulate in the seabed sediment and affect subtidal benthic invertebrates through the introduction of excess nutrients and toxic substances (Islam & Tanaka 2004). The use, dosage and disposal of agrichemicals can potentially be regulated with the aim of reducing their environmental impacts. This would likely limit or reduce the amount that enters the marine environment and reduce the risks to subtidal benthic invertebrates associated with this pollution.

Evidence for other interventions related to pollution from agriculture are summarised under "Threat: Pollution – Treat wastewater from intensive livestock holdings", "Establish aquaculture to extract the nutrients from run-offs" and "Create artificial wetlands to reduce the amount of pollutants reaching the sea".

Islam S. & Tanaka M. (2004) Impacts of pollution on coastal and marine ecosystems including coastal and marine fisheries and approach for management: a review and synthesis. *Marine Pollution Bulletin,* 48, 624–649.

Rawlins B.G., Ferguson A.J., Chilton P.J., Arthurton R.S., Rees J.G. & Baldock J.W. (1998) Review of agricultural pollution in the Caribbean with particular emphasis on small island developing states. *Marine Pollution Bulletin*, 36, 658–668.

9.25. Treat wastewater from intensive livestock holdings

- We found no studies that evaluated the effects of treating wastewater from intensive livestock holdings on subtidal benthic invertebrate 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

Intensive agriculture constitutes an important source of pollution to the marine environment. Agriculture waste and pollutants can enter rivers and other watercourse runoffs and be discharged into the sea (Rawlings *et al.* 1998). For instance, wastewater from intensive livestock holdings can introduce bacteria, excess nutrients and solid particles. Treating wastewater from intensive livestock holdings may reduce pollution levels in the marine environment and reduce associated risks to subtidal benthic invertebrates.

Evidence for other interventions related to pollution from agriculture are summarised under "Threat: Pollution – Regulate the use, dosage and disposal of agrichemicals", "Establish aquaculture to extract the nutrients from run-offs" and "Create artificial wetlands to reduce the amount of pollutants reaching the sea".

Rawlins B.G., Ferguson A.J., Chilton P.J., Arthurton R.S., Rees J.G. & Baldock J.W. (1998) Review of agricultural pollution in the Caribbean with particular emphasis on small island developing states. *Marine Pollution Bulletin*, 36, 658–668.

9.26. Establish aquaculture to extract the nutrients from run-offs

- We found no studies that evaluated the effects of establishing aquaculture to extract the nutrients from run-offs on subtidal benthic invertebrate 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

Intensive agriculture constitutes an important source of pollution to the marine environment. Agriculture waste and pollutants can enter rivers and other watercourse runoffs and be discharged into the sea (Rawlings *et al.* 1998). For instance, wastewaters can introduce agrichemicals, bacteria, excess nutrients and solid particles, which negatively impact on subtidal benthic invertebrates (Rawlings *et al.* 1998). Some species can naturally improve water quality through feeding (for instance filter-feeding species such as mussels), or through photosynthesis (for instance algae species) (Chung *et al*. 2002). Establishing aquaculture systems near polluted areas from agriculture runoffs and wastewaters in order to extract nutrients may be an effective biological method of pollution removal ('bioemediation'; Desilva *et al*. 2000), which may reduce pollution levels and allow subtidal benthic invertebrate communities to recover over time.

Evidence for other interventions related to pollution from agriculture are summarised under "Threat: Pollution – Regulate the use, dosage and disposal of agrichemicals", "Treat wastewater from intensive livestock holdings", and "Create artificial wetlands to reduce the amount of pollutants reaching the sea".

Chung I.K., Kang Y.H., Yarish C., Kraemer G.P. & Lee J.A. (2002) Application of seaweed cultivation to the bioremediation of nutrient-rich effluent. *Algae,* 17, 187–194.
Desilva S., Ingram B.A., Gooley G.F. & McKinon L.J. (2000) Aquaculture-agriculture systems integration: an Australian perspective. *Fisheries Management and Ecology*.
Rawlins B.G., Ferguson A.J., Chilton P.J., Arthurton R.S., Rees J.G. & Baldock J.W. (1998) Review of agricultural pollution in the Caribbean with particular emphasis on small island developing states. *Marine Pollution Bulletin*, 36, 658–668.

9.27. Create artificial wetlands to reduce the amount of pollutants reaching the sea

• We found no studies that evaluated the effects of creating artificial wetlands to reduce the amount of pollutants reaching the sea on subtidal benthic invertebrate 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

Intensive agriculture constitutes an important source of pollution to the marine environment. Agriculture waste and pollutants can enter rivers and other watercourse runoffs and be discharged into the sea (Rawlings *et al.* 1998). For instance, wastewaters can introduce agrichemicals, bacteria, excess nutrients and solid particles, which negatively impact on subtidal benthic invertebrates (Rawlings *et al.* 1998). Artificial wetlands can be created with the aim of retaining agricultural pollution (Smiley & Alfred 2011). For instance, solid particles sink in areas of slow water flow and plants growing on the wetlands can remove excess nutrients (Brix 1994). Creating artificial wetlands near agricultural lands may reduce the amount of pollutants reaching the marine environment and reduce associated risks to subtidal benthic invertebrates.

Evidence for other interventions related to pollution from agriculture are summarised under "Threat: Pollution – Regulate the use, dosage and disposal of agrichemicals", Treat wastewater from intensive livestock holdings", and "Establish aquaculture to extract the nutrients from run-offs"

Brix H. (1994) Use of constructed wetlands in water pollution control: historical development, present status, and future perspectives. *Water Science and Technology,* 30, 209–223.
Rawlins B.G., Ferguson A.J., Chilton P.J., Arthurton R.S., Rees J.G. & Baldock J.W. (1998) Review of agricultural pollution in the Caribbean with particular emphasis on small island developing states. *Marine Pollution Bulletin*, 36, 658–668.
Smiley P.C. & Allred B.J. (2011) Differences in aquatic communities between wetlands created by an agricultural water recycling system. *Wetlands Ecology and Management*, 19, 495–505.

Garbage and solid waste

9.28. Limit, cease or prohibit discharge of solid waste overboard from vessels

• We found no studies that evaluated the effects of limiting, ceasing or prohibiting discharge of solid waste overboard from vessels on subtidal benthic invertebrate 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

Commercial and recreational vessels can generate large amounts of garbage and solid waste (Butt 2007). Wastes discharged overboard from vessels can sink to the seabed and impact subtidal benthic invertebrates through the introduction of bacteria, excess nutrients, toxic substances and solid particles. Limiting, ceasing or prohibiting the discharge of waste overboard from vessels in an area may reduce or stop the source of pollution and allow subtidal benthic invertebrates to recover over time. However, solid waste can accumulate and subsist in the marine environment for a long time due to very slow degradation (Andrady 2015; Pham *et al*. 2014), and limiting, ceasing or prohibiting discharge alone might not be sufficient.

Evidence for intervention related to the discharge of waste effluents is summarised under "Threat: Pollution – Limit, cease or prohibit discharge of waste effluents overboard from vessels".

Andrady A.L. (2015) Persistence of plastic litter in the oceans. Pages 57–72 in: *Marine anthropogenic litter*. Springer, Cham.

Butt N. (2007) The impact of cruise ship generated waste on home ports and ports of call: A study of Southampton. *Marine Policy*, 31, 591–598.

Pham C.K., Ramirez-Llodra E., Alt C.H., Amaro T., Bergmann M., Canals M., Davies J., Duineveld G., Galgani F., Howell K.L. & Huvenne V.A. (2014) Marine litter distribution and density in European seas, from the shelves to deep basins. *PloS One*, 9, p.e95839.

9.29. Install stormwater traps or grids

- We found no studies that evaluated the effects of installing stormwater traps or grids on subtidal benthic invertebrate 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

Litter can enter the marine environment through a multitude of pathways. Urban debris can enter the marine environment in unprocessed stormwaters running off from land via stormwater conduits and drainage systems (Armitage & Rooseboom 2000). Once in the marine environment, litter can accumulate and subsist for a long time due to very slow degradation (Andrady 2015). Litter can negatively affect subtidal benthic invertebrates through physical damage, smothering and habitat modification, but also through the introduction of bacteria, nutrients, toxic substances and other solid particles (Gall & Thompson 2015). Stormwater traps or grids are designed to prevent litter from entering stormwaters (Armitage & Rooseboom 2000; Phillips 1999). Their installation can potentially reduce the amount of urban litter entering the marine environment, and reduce the risks associated with this pollution on subtidal benthic invertebrates.

Evidence for intervention related to pollution from sewage system is summarised under "Threat: Pollution – Set or improve minimum sewage treatment standards", and "Limit the amount of storm wastewater overflow".

Andrady A.L. (2015) Persistence of plastic litter in the oceans. Pages 57–72 in: *Marine anthropogenic litter*. Springer, Cham.
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.
Phillips D.I. (1999). A new litter trap for urban drainage systems. *Water Science and Technology*, 39, 85–92.

9.30. Remove litter from the marine environment

• We found no studies that evaluated the effects of removing litter from the marine environment on subtidal benthic invertebrate 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

Litter can enter the marine environment through a multitude of pathways, for instance vessels, rivers, storms, beaches, fishing activities. Once in the marine environment, litter can accumulate and subsist for a long time due to very slow degradation (Andrady 2015; Pham *et al*. 2014). Litter can negatively affect subtidal benthic invertebrates through physical damage, smothering and habitat modification, but also through the introduction of bacteria, nutrients, toxic substances and other solid particles (Gall & Thompson 2015).

Removing litter from the marine environment may temporarily remove the cause of harm and allow subtidal benthic invertebrates to recover. However, this intervention is only deals with the physical pollution, but not with its source and cause, and therefore can only be considered temporary.

Evidence of interventions related to the source of litter and solid pollution are summarised under "Threat: Pollution – Limit, cease or prohibit discharge of solid waste overboard from vessels" and "Install stormwater traps or grids". Evidence for other interventions related to the removal of solid litter from the marine environment are summarised under "Threat: Pollution – Recover lost fishing gear" and "Remove and clean-up shoreline waste disposal sites".

Andrady A.L. (2015) Persistence of plastic litter in the oceans. Pages 57–72 in: *Marine anthropogenic litter*. Springer, Cham.
Gall S.C. & Thompson R.C. (2015) The impact of debris on marine life. *Marine Pollution Bulletin*, 92, 170–179.
Pham C.K., Ramirez-Llodra E., Alt C.H., Amaro T., Bergmann M., Canals M., Davies J., Duineveld G., Galgani F., Howell K.L. & Huvenne V.A. (2014) Marine litter distribution and density in European seas, from the shelves to deep basins. *PloS One*, 9, p.e95839.

9.31. Use biodegradable panels in fishing pots

• We found no studies that evaluated the effects of using biodegradable panels in fishing pots on subtidal benthic invertebrate 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

Fishing is an important source of marine litter (Matsuoka *et al*. 2005). Once in the marine environment, discarded or lost fishing gears, often referred to as '*ghost fishing*' (Matsuoka *et al*. 2005), can accumulate and subsist for a long time due to very slow degradation. There, it can harm subtidal benthic invertebrates through physical damage, entanglement, smothering and habitat modification (Gilardi *et al.* 2010). Biodegradable panels can be used in fishing pots so that they degrade over time in the marine environment, without harmful effects (Bilkovic *et al.* 2012). By using biodegradable panels on fishing pots, the risk of subtidal benthic invertebrates getting caught in lost or discarded pots is potentially reduced, or if they are caught, they may be able to escape as the pots degrade.

Evidence for interventions related to pollution from fishing gear are summarised under "Threat: Pollution – Recover lost fishing gear".

Bilkovic D.M., Havens K.J., Stanhope D.M. & Angstadt K.T. (2012) Use of fully biodegradable panels to reduce derelict pot threats to marine fauna. *Conservation Biology,* 26, 957–966.

Gilardi K.V., Carlson-Bremer D., June J.A., Antonelis K., Broadhurst G.,& Cowan T. (2010) Marine species mortality in derelict fishing nets in Puget Sound, WA and the cost/benefits of derelict net removal. *Marine Pollution Bulletin*, 60, 376–382.

Matsuoka T., Nakashima T. & Nagasawa N. (2005) A review of ghost fishing: scientific approaches to evaluation and solutions. *Fisheries Science*, 71, 691.

9.32. Recover lost fishing gear

- We found no studies that evaluated the effects of recovering lost fishing gear on subtidal benthic invertebrate 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

Fishing is an important source of marine litter (Gall & Thompson 2015; Matsuoka *et al*. 2005). Once in the marine environment, discarded or lost fishing gears, often referred to as '*ghost fishing*' (Matsuoka *et al*. 2005), can accumulate and subsist for a long time due to very slow degradation. There, it can harm subtidal benthic invertebrates through physical damage, entanglement, smothering and habitat modification (Gall & Thompson 2015; Gilardi *et al.* 2010). 'Ghost fishing' gear, once located, can be recovered and removed from the marine environment, which can reduce the risk of subtidal benthic invertebrates getting caught or harmed (Gilardi *et al.* 2010).

Evidence for interventions related to pollution from fishing gear are summarised under "Threat: Pollution – Use biodegradable panels in fishing pots". Evidence for other interventions related to the removal of solid litter from the marine environment are summarised under "Threat: Pollution – Remove litter from the marine environment" and "Remove and clean-up shoreline waste disposal sites".

Gall S.C. & Thompson R.C. (2015) The impact of debris on marine life. *Marine Pollution Bulletin*, 92, 170–179

Gilardi K.V., Carlson-Bremer D., June J.A., Antonelis K., Broadhurst G. & Cowan T. (2010) Marine species mortality in derelict fishing nets in Puget Sound, WA and the cost/benefits of derelict net removal. *Marine Pollution Bulletin*, 60, 376–382.
Matsuoka T., Nakashima T. & Nagasawa N. (2005) A review of ghost fishing: scientific approaches to evaluation and solutions. *Fisheries Science*, 71, 691.

Excess energy

Light and Noise pollution

9.33. Bury electricity cables to reduce electromagnetic fields

• We found no studies that evaluated the effects of burying electricity cables to reduce electromagnetic fields on subtidal benthic invertebrate 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

Emission of electromagnetic fields is associated with electricity production during the operational phase of offshore renewable energy installations (Gill 2005; Inger *et al*. 2009). In particular, the electromagnetic fields emitted by subsea cables (transmitting power between devices and the mainland) have been shown to affect benthic species sensitive to electromagnetic fields, including subtidal invertebrates such as the edible crab *Cancer pagurus* (Scott *et al.* 2019) and the American lobster, *Hommarus americanus* (Hutchison *et al*. 2018). Industry standards and good practice require that all subsea cables in the water of up to 1,500 m should be buried in the seabed (Carter *et al.* 2009). Burying subsea cables in an area might help reduce the effects of electromagnetic fields on subtidal benthic invertebrates.

Carter L., Burnett D., Drew S., Marle G., Hagadorn L., Bartlett McNeil D. & Irvine N. (2009) *Submarine cables and the oceans – connecting the world.* UNEP-WCMC Biodiversity Series No. 31. ICPC/UNEP/UNEP-WCMC.
Gill A.B. (2005) Offshore renewable energy: ecological implications of generating electricity in the coastal zone. *Journal of Applied Ecology*, 42, 605–615.
Hutchison Z.L., Sigray P., He H., Gill A.B., King J. & Gibson C. (2018) *Electromagnetic Field (EMF) impacts on elasmobranch (shark, rays, and skates) and American lobster movement and migration from direct current cables*. Sterling (VA): US Department of the Interior, Bureau of Ocean Energy Management. OCS Study BOEM, 3.
Inger R., Attrill M.J., Bearhop S., Broderick A.C., Grecian W.J., Hodgson D.J., Mills C., Sheehan E., Votier S.C., Witt M.J. & Godley B.J. (2009) Marine renewable energy: potential benefits to biodiversity? An urgent call for research. *Journal of Applied Ecology*, 46,1145–1153.
Scott K., Harsanyi P. & Lyndon A.R. (2019) Understanding the effects of electromagnetic field emissions from Marine Renewable Energy Devices (MREDs) on the commercially important edible crab, *Cancer pagurus* (L.). *Frontier in Marine Science Conference Abstract: IMMR'18 | International Meeting on Marine Research 2018.*

9.34. Limit, cease or prohibit industrial and urban lighting at night

• We found no studies that evaluated the effects of limiting, ceasing or prohibiting industrial and urban lighting at night on subtidal benthic invertebrate 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

Artificial lighting at night, including urban lighting from street lights and houses, industrial lighting from shops and offices, and underwater lighting in marinas and on pontoons, has recently been recognized as a cause for marine environmental concern (Davies *et al.* 2014). Light pollution has been shown to negatively affect the behaviour of several species, and the community composition of marine invertebrates (Davies *et al.* 2015; Navarro-Barranco & Hughes 2015). Limiting (in duration, intensity or spectral composition), ceasing or prohibiting lighting at night in an area may reduce the levels of light pollution affecting subtidal benthic invertebrates (Gaston *et al.* 2012).

Davies T.W., Coleman M., Griffith K.M. & Jenkins S.R. (2015) Night-time lighting alters the composition of marine epifaunal communities. *Biology Letters*, 11, 20150080.

Davies T.W., Duffy J.P., Bennie J. & Gaston K.J. (2014) The nature, extent, and ecological implications of marine light pollution. *Frontiers in Ecology and the Environment*, 12, 347–355.

Gaston K.J., Davies T.W., Bennie J. & Hopkins J. (2012) Reducing the ecological consequences of night-time light pollution: options and developments. *Journal of Applied Ecology*, 49, 1256–1266.

Navarro-Barranco C. & Hughes L.E. (2015) Effects of light pollution on the emergent fauna of shallow marine ecosystems: Amphipods as a case study. *Marine Pollution Bulletin*, 94, 235–240.

9.35. Reduce underwater noise (other than sonar)

- We found no studies that evaluated the effects of reducing underwater noise on subtidal benthic invertebrate 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 subtidal benthic invertebrate species rely on sound to communicate, feed, navigate, detect predators, and reproduce (Popper & Hawkins 2012). Underwater noise, for instance from shipping, fishing, construction work, or aggregate extraction can mask the ambient soundscape and negatively affect subtidal benthic invertebrate species that rely on it (de Soto 2016; Pine *et al*. 2012; Popper & Hawkins 2012). Reducing underwater noise, for instance by limiting the level, intensity, and duration of specific noise-generating activities, or by using 'technologies' to dampen underwater noise emissions (such as bubble curtains, hydro-sound dampeners), may prevent the negative effects on subtidal benthic invertebrates.

Evidence for interventions related to noise pollution from sonars are summarised under "Threat: Pollution – Limit, cease or prohibit the use of sonars".

de Soto N.A. (2016) Peer-reviewed studies on the effects of anthropogenic noise on marine invertebrates: from scallop larvae to giant squid. Pages 17–26 in: *The Effects of Noise on Aquatic Life II*. Springer, New York, NY.

Pine M.K., Andrew J.G. & Radford C.A. (2012) Turbine sound may influence the metamorphosis behaviour of estuarine crab megalopae. *PLoS One*, 7, p. e51790

Popper A.N. & Hawkins A. (Eds.) (2012) *The Effects of Noise on Aquatic Life*, Springer Science + Business Media, LLC, New York

9.36. Limit, cease or prohibit the use of sonars

- We found no studies that evaluated the effects of limiting, ceasing or prohibiting the use of sonars on subtidal benthic invertebrate 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 subtidal benthic invertebrate species rely on sound to communicate, feed, navigate, detect predators, and reproduce. Underwater sonar sounds, for instance from naval ships, may be perceived by marine animals as a threat and disturb their anti-predator responses (Abate 2010; Harris *et al*. 2018). While the threats associated with sonar are so far mainly linked with marine mammals, in contrast little to no research has been undertaken on marine subtidal benthic invertebrates. However, this does not mean the use of sonar does not represent a potential threat to them, particularly as they can be negatively affected by other underwater anthropogenic noise (de Soto 2016; Pine *et al.* 2012). Limiting, ceasing or prohibiting the use of sonar underwater could prevent potential unforeseen responses from subtidal benthic invertebrates.

Abate R.S. (2010) NEPA, national security, and ocean noise: the past, present, and future of regulating the impact of navy sonar on marine mammals. *Journal of International Wildlife Law & Policy*, 13, 326–356.
Harris C.M., Thomas L., Falcone E.A., Hildebrand J., Houser D., Kvadsheim P.H., Lam F.P.A., Miller P.J., Moretti D.J., Read A.J. & Slabbekoorn H. (2018) Marine mammals and sonar: Dose-response studies, the risk-disturbance hypothesis and the role of exposure context. *Journal of Applied Ecology*, 55, 396–404.
de Soto N.A. (2016) Peer-reviewed studies on the effects of anthropogenic noise on marine invertebrates: from scallop larvae to giant squid. Pages 17–26 in: *The Effects of Noise on Aquatic Life II*. Springer, New York, NY.
Pine M.K., Andrew J.G. & Radford C.A. (2012) Turbine sound may influence the metamorphosis behaviour of estuarine crab megalopae. *PLoS One*, 7, p. e51790

Thermal pollution

9.37. Limit, cease or prohibit the discharge of cooling effluents from power stations

- We found no studies that evaluated the effects of limiting, ceasing or prohibiting the discharge of cooling effluents from power stations on subtidal benthic invertebrate 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

The cooling water effluents of power stations represent a localized discharge of water 8-12°C above ambient seawater (Bamber 1990), often containing chemicals, and constitute a well-established threat to marine organisms (Barnett 1972; Naylor 1965). Limiting, ceasing or prohibiting the discharge of cooling effluents from power stations in an area can potentially reduce or stop the source of pollution and allow subtidal benthic invertebrates to recover over time.

Bamber R.N. (1990) Power station thermal effluents and marine crustaceans. *Journal of Thermal Biology*, 15, 91–96.
Barnett P.R.O. (1972). Effects of warm water effluents from power stations on marine life. *Proceedings of the Royal Society of London: B,* 180, 497–509.
Naylor E. (1965) Effects of heated effluents upon marine and estuarine organisms. Pages 63–103 in: *Advances in Marine Biology*. Academic Press.

Other pollution

9.38. Limit, cease or prohibit the discharge of waste effluents overboard from vessels

- We found no studies that evaluated the effects of limiting, ceasing or prohibiting the discharge of waste effluents overboard from vessels on subtidal benthic invertebrate 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

Commercial, recreational, industrial, and military vessels can generate large amounts of liquid waste, such as sewage, grey waters, and bilge waters (Welles 2003). Discharge of these wastes overboard from vessels can impact subtidal benthic invertebrates through the introduction of bacteria, excess nutrients, toxic substances and solid particles. Limiting, ceasing or prohibiting the discharge of waste overboard from vessels in an area can potentially reduce or stop the source of pollution and allow subtidal benthic invertebrates to recover over time. In many parts of the world, it is illegal to dispose of waste effluents into coastal waters or delimited zones, for instance following local bylaws.

Evidence for interventions related to the discharge of solid wastes overboard are summarised under "Threat: Pollution – Limit, cease or prohibit the discharge of solid waste overboard from vessels".

Welles L.K. (2003) Comment: Due to loopholes in the Clean Water Act, what can a state do to combat cruise ship discharge of sewage and gray water. *Ocean & Coastal Law Journal*, 9, 99.

9.39. Use non-toxic antifouling coatings on surfaces

- We found no studies that evaluated the effects of using non-toxic antifouling coatings on surfaces on subtidal benthic invertebrate 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

Antifouling paints and coatings are commonly used to manage *biofouling* (organisms that can attach to hard surfaces) on aquaculture structures (cages, nets, ponds) and other hard anthropogenic structures. However, some antifouling paints and coatings are highly toxic to marine organisms. Trybutyltin (TBT) for instance was widely used on vessels but found to be highly harmful to marine invertebrates, such as the dog whelk *Nucella lapillus* (Alzieu 2000; Gibbs *et al*. 2009) and so the use of this substance was banned (for evidence on the restriction of TBT, see "Threat: Pollution – Restrict the use of tributyltin or other toxic antifouling coatings"). Using non-toxic antifouling coatings instead of more traditional coatings may reduce the risk of toxicity to subtidal benthic invertebrates (Magin *et al*. 2010).

Alzieu C. (2000) Impact of tributyltin on marine invertebrates. *Ecotoxicology*, 9, 71-76.

Gibbs P.E., Bryan G.W., Pascoe P.L. & Burt G.R. (2009) The use of the dog-whelk, *Nucella lapillus*, as an indicator of tributyltin (TBT) contamination. *Journal of the Marine Biological Association of the United Kingdom,* 67, 507.

Magin C.M., Cooper S.P. & Brennan A.B. (2010). Non-toxic antifouling strategies. *Materials Today*, 13, 36-44.

9.40. Restrict the use of tributyltin or other toxic antifouling coatings

- **Four studies** examined the effects of restricting the use of tributyltin as an antifouling coating on subtidal benthic invertebrate populations. One study was located in the English Channel[1] (UK), two in the River Crouch estuary[2,3] (UK), and one in Otsuchi Bay[4] (Japan).

COMMUNITY RESPONSE (2 STUDIES)

- **Overall community composition (1 study):** One replicated, before-and-after study in the River Crouch estuary[2] found that after restricting the use of tributyltin, invertebrate community composition changed, but that changes varied with locations.
- **Overall richness/diversity (1 study):** One replicated, before-and-after study in the River Crouch estuary[3] found that after restricting the use of tributyltin, overall invertebrate species richness and diversity increased.

POPULATION RESPONSE (2 STUDIES)

- **Molluscs condition (1 study):** One replicated, before-and-after study in the English Channel[1] found that after restricting the use of tributyltin, there was a decrease in its concentration in dogwhelks and the penis length of female dogwhelks.
- **Crustacean condition (1 study):** One study in Otsuchi Bay[4] found that after restricting the use of tributyltin its concentration decreased in skeleton shrimps.

Background

Antifouling paints and coatings are commonly used to manage *biofouling* (organisms that can attach to hard surfaces) on vessels and other hard anthropogenic structures. However, some antifouling paints and coatings are highly harmful and toxic to marine organisms (Roach & Wilson 2009). Trybutyltin (TBT) for instance was widely used on vessels but found to be very harmful to marine invertebrates, such as the dog whelk *Nucella lapillus* whose females developed male genitals (Alzieu 2000; Gibbs *et al*. 2009), and a ban on the use of this substance was then instated. Restricting the use of tributyltin or other toxic antifouling coatings can reduce or entirely remove the source of toxicity over time from the seawater, therefore reducing the risk of harm to subtidal benthic invertebrates and allowing populations to recover (Bryan *et al*. 1993). Related evidence for the use of antifouling coatings are summarised under "Threat: Pollution – Use non-toxic antifouling coatings on surfaces".

Alzieu C. (2000) Impact of tributyltin on marine invertebrates. *Ecotoxicology*, 9, 71–76.

Bryan G.W., Burt G.R., Gibbs P.E. & Pascoe P.L. (1993) *Nassarius reticulatus* (Nassariidae: Gastropoda) as an indicator of tributyltin pollution before and after TBT restrictions. *Journal of the Marine Biological Association of the United Kingdom*, 73, 913–929.

Gibbs P.E., Bryan G.W., Pascoe P.L. & Burt G.R. (2009) The use of the dog-whelk, *Nucella lapillus*, as an indicator of tributyltin (TBT) contamination. *Journal of the Marine Biological Association of the United Kingdom*, 67, 507.

Roach A.C. & Wilson S.P. (2009). Ecological impacts of tributyltin on estuarine communities in the Hastings River, NSW Australia. *Marine Pollution Bulletin*, 58, 1780–1786.

A replicated, before-and-after study in 1985–1993 of seven soft seabed sites in the southwest English Channel, UK (1) found that restricting the use of tributyltin (TBT) resulted in decreases in tributyltin concentrations in netted dogwhelks *Nassarius reticulatus* and in the penis length of female dogwhelks, at five of the sites five years after the restriction. In these five sites, TBT concentrations in dogwhelks were lower five years after the restriction (8–68 ng/g; range of averages across sites) compared to before (123–390 ng/g). The penis length of females was lower five years after the restriction (3–5 mm) compared to before (5–7 mm). There were no changes in the penis length of males over time (data not presented). There were no statistical trends over time at the other two sites. The use of antifouling ship paints containing tributyltin was restricted in 1987 in the UK. Approximately every six months between 1985 and 1993, at least 30–40 dogwhelks were collected using a dredge or by diving at each of seven sites at 9 m depth. The TBT concentration of each individual (after being dried) and their penis length were measured as affected female marine invertebrates develop male genitals.

A replicated, before-and-after study in 1987–1992 of seven coarse seabed sites along the River Crouch estuary, southeast England, UK (2 – same experimental set-up as 3) found that after restricting the use of tributyltin (TBT), overall epifaunal invertebrate (living on the seabed) community composition changed over five years, but the direction of change varied with site location. In four upper-estuary sites, overall invertebrate community composition changed over the five years, in a similar direction. In three lower-estuary sites, overall invertebrate community composition changed over the five years, without displaying a directional trend. Data were reported as graphical analyses, but not statistically tested. The use of antifouling ship paints containing TBT was restricted in 1987 in the UK. Annually in 1987–1989 and 1992, epifaunal invertebrates were surveyed at seven sites along a 23 km axis of the river. One to three sediment samples/year/site were collected using a trawl towed over 250 m, epifaunal invertebrates (> 5 mm) were identified and counted.

A replicated, before-and-after study in 1987–1991 of five soft seabed sites along the River Crouch estuary, southeast England, UK (3 – same experimental set-up as 2) found that after restricting the use of tributyltin (TBT), infaunal invertebrate (living inside the seabed) species richness increased at all sites and diversity increased at sites located in the upper-estuary over four years. Data were not statistically tested. Species richness increased from 37 species before restriction to 63 four years after. Upper-estuary sites showed the greatest increases from 5–7 to 19–26 species, while the lower-estuary sites varied from 9–12 species before to 15–22 after. Before restriction, species diversity (reported as a diversity index) was higher at lower-estuary sites than upper-estuary sites.

After restriction, diversity remained similar at lower-estuary sites but increased at upper-estuary sites so that they reached similar values to the lower estuary sites. TBT concentrations in sediments, although higher in the upper estuary than the lower estuary, decreased over time at all sites. The use of antifouling ship paints containing TBT was restricted in 1987 in the UK. Annually in 1987–1988 and 1990–1991, infaunal invertebrates (> 5 mm) were surveyed at five sites along the length of the estuary. Four sediment samples/year/site were collected using a grab, and invertebrates identified and counted.

A study in 1994–2001 of one soft seabed sites in Otsuchi Bay, northern Japan (4) found that, between four and 11 years after restricting its use, tributyltin (TBT) was still present in four species of *Caprella* skeleton shrimps, but concentrations were declining. Tributyltin concentrations significantly declined in *Caprella danilevskii* from 59 ng/g four years after restriction to 3.3 ng/g 11 years after restriction, in *Caprella subinermis* from 57 ng/g in four years after restriction to 29 ng/g 10 years after restriction, in *Caprella penantis* from 66 ng/g five years after restriction to 4 ng/g 11 years after restriction, and in *Caprella verrucosa* from 32 ng/g seven years after restriction to 10 ng/g nine years after restriction. The use of antifouling ship paints containing tributyltin was restricted in 1990 in Japan. Annually in 1994–2001 an unspecified number of shrimps living on the macroalgae *Sargassum* were collected at one site (3 m depth). They were then identified as one of four *Caprella* species, and tribultyltin concentrations measured for each species. Not all species were collected each year.

(1) Bryan G.W., Burt G.R., Gibbs P.E. & Pascoe P.L. (1993) *Nassarius reticulatus* (Nassariidae: Gastropoda) as an indicator of tributyltin pollution before and after TBT restrictions. *Journal of the Marine Biological Association of the United Kingdom*, 73, 913–929.
(2) Rees H.L., Waldock R., Matthiessen P. & Pendle M.A. (1999) Surveys of the epibenthos of the Crouch Estuary (UK) in relation to TBT contamination. *Journal of the Marine Biological Association of the United Kingdom*, 79, 209–223.
(3) Waldock R., Rees H.L., Matthiessen P. & Pendle M.A. (1999) Surveys of the benthic infauna of the Crouch Estuary (UK) in relation to TBT contamination. *Journal of the Marine Biological Association of the United Kingdom*, 79, 225–232.
(4) Takeuchi I., Takahashi S. & Tanabe S. (2004) Decline of butyltin levels in *Caprella* spp. (Crustacea: Amphipoda) inhabiting the *Sargassum* community in Otsuchi Bay, Japan from 1994 to 2001. *Journal of the Marine Biological Association of the United Kingdom*, 84, 911–918.

9.41. Remove and clean-up shoreline waste disposal sites

- **One study** examined the effects of removing and cleaning-up shoreline waste disposal sites on subtidal benthic invertebrates. The study was in the Southern Ocean[1] (Antarctica).

COMMUNITY RESPONSE (1 STUDY)

- **Overall community composition (1 study):** One replicated, controlled, before-and-after study in the Southern Ocean[1] found that after removing and cleaning-up a disused waste disposal site, invertebrate community composition changed, and no further negative impacts were detected, but communities remained different to natural sites.
- **Overall richness/diversity (1 study):** One replicated, controlled, before-and-after study in the Southern Ocean[1] found that after removing and cleaning-up a disused waste disposal site, invertebrate species richness did not change over time and remained different to that of natural sites, but no further negative impacts were detected.

POPULATION RESPONSE (0 STUDIES)

Background

In parts of the world, such as Antarctica, waste has been dumped in landfill sites (and onto sea ice in Antarctica) for lack of better solutions (Stark *et al*. 2006). The waste disposal sites can be highly contaminated, and when occurring near the coastal zone can negatively affect marine subtidal benthic invertebrates. Removing and cleaning shoreline waste disposal sites can remove the direct source of pollution and threat, and benefit organisms who may naturally recolonize or recover over time (Stark *et al.* 2014).

Stark J.S., Johnstone G.J. & Riddle M.J. (2014) A sediment mesocosm experiment to determine if the remediation of a shoreline waste disposal site in Antarctica caused further environmental impacts. *Marine Pollution Bulletin*, 89, 284–295.

Stark J.S., Snape I. & Riddle M.J. (2006) Abandoned Antarctic waste disposal sites: monitoring remediation outcomes and limitations at Casey Station. *Ecological Management & Restoration*, 7, 21–31.

A replicated, controlled, before-and-after study in 2001–2006 of four sites off Casey Station, Southern Ocean, East Antarctica (1) found that over the two years after cleaning-up a shoreline waste disposal site, invertebrate community compositions at two adjacent impacted subtidal sites changed but remained different to that of two further afield natural subtidal sites. However, no additional negative impacts were detected. Invertebrate communities were significantly different at the impacted sites compared to the natural sites, both before and after removal, and changes over time were similar at impacted and natural sites (data reported as graphical analyses). In addition, species richness did not decrease over time at the impacted sites (before: 13–15; after: 12–18 species/sample), and after two years remained lower than at the natural sites (impacted: 16–18; natural: 20–22 species/sample). In 2003–2004, a disused waste disposal site of an Antarctic research station was removed and cleaned-up to comply with the Antarctic Treaty. Two impacted sites (50 and 200 m from the disposal site) and two nearby natural sites (>2 km away) were monitored. Four groups of five trays (34 x 23 x 12 cm; 20 m between groups) filled with sediments without invertebrates were deployed at 7–15 m depth at each site. One year before, one month before, one month after, and two years after the clean-up, invertebrates were sampled from one tray/group/site using a core (10 cm diameter) and extracted (methodology unspecified).

(1) Stark J.S., Johnstone G.J. & Riddle M.J. (2014) A sediment mesocosm experiment to determine if the remediation of a shoreline waste disposal site in Antarctica caused further environmental impacts. *Marine Pollution Bulletin*, 89, 284–295.

10. Threat: Climate change and severe weather

Background:

Climate change and extreme weather are expected to severely alter global marine biodiversity (Cheung *et al.* 2009). However, they are very large-scale threats, and therefore most interventions that could be used in response to them are general conservation interventions discussed in other chapters, such as restoring habitats and translocating species (discussed in "Habitat restoration and creation" and "Species management", respectively). The literature on the effects of climate change and related threats to the marine environment, including ocean acidification, hypoxia, and salinity changes, is relatively recent (Harley *et al.* 2006; Hoegh-Guldberg & Bruno 2010), and as such little conservation is undertaken to pre-emptively counteract their effects. There is however a growing body of literature that is laboratory-based, particularly regarding genetically modifying species and population to enhance resilience and resistance; but this is not discussed in this synopsis which focusses on *in situ* conservation actions.

Cheung W.W., Lam V.W., Sarmiento J.L., Kearney K., Watson R. & Pauly D. (2009) Projecting global marine biodiversity impacts under climate change scenarios. *Fish and Fisheries*, 10, 235–251.

Harley C.D., Randall Hughes A., Hultgren K.M., Miner B.G., Sorte C.J., Thornber C.S., Rodriguez L.F., Tomanek L. & Williams S.L. (2006) The impacts of climate change in coastal marine systems. *Ecology Letters*, 9, 228–241.

Hoegh-Guldberg O. & Bruno J.F. (2010) The impact of climate change on the world's marine ecosystems. *Science*, 328, 1523–1528.

10.1. Restore habitats and/or habitat-forming (biogenic) species following extreme events

- We found no studies that evaluated the effects of restoring habitats and/or habitat-forming species following extreme events on subtidal benthic invertebrate 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

Extreme events, such as short periods of unusually high or low temperatures, heavy flooding episodes, or intense storms with heavy rainfall, strong winds and scouring, can profoundly impact the marine environment through alterations in environmental conditions and sediment type (Smale & Wernberg 2013). Extreme events are expected to intensify with on-going climate change (Easterling *et al.* 2000), with likely negative consequences for subtidal benthic invertebrates. For instance, heat waves and flooding episodes have been associated with declines in benthic communities, including bivalves, crustaceans, and worms in Portugal (Grillo *et al.* 2011). Restoring habitats and/or habitat-forming (biogenic) species following extreme events may potentially help subtidal benthic invertebrate species recover following the disturbances.

Interventions aimed at restoring habitats and/or habitat-forming species, outside of the context of climate change, are listed in the following chapters: "Species management" and "Habitat restoration and creation".

Easterling D.R., Meehl G.A., Parmesan C., Changnon S.A., Karl T.R. & Mearns L.O. (2000) Climate extremes: observations, modeling, and impacts. *Science* 289, 2068–2074.
Grilo T.F., Cardoso P.G., Dolbeth M., Bordalo M.D. & Pardal M.A. (2011) Effects of extreme climate events on the macrobenthic communities' structure and functioning of a temperate estuary. *Marine Pollution Bulletin*, 62, 303–311.
Smale D.A. & Wernberg T. (2013. Extreme climatic event drives range contraction of a habitat-forming species. *Proceedings of the Royal Society B: Biological Sciences*, 280, 1754.

10.2. Manage climate-driven range extensions of problematic species

- We found no studies that evaluated the effects of managing climate-driven range extensions of problematic species on subtidal benthic invertebrate 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

Range extensions, where the geographic area occupied by a species naturally (without human interference) changes over time, can occur in response to climate change and extreme events (Kyle *et al.* 2014; Pitt *et al.* 2010). In some cases, the 'new' species to the area (following range extension) can become problematic and negatively impact on other marine species (Johnson *et al.* 2011). For instance, the range of the barren-forming sea urchin *Centrostephanus rodgersii* has extended poleward from New South Wales to Tasmania over the past 40 years or so, and has been shown to be a contributing factor to the cascading negative ecological effect on the local rocky reef community, shifting from macroalgae-dominated systems to sea urchin barrens (Johnson *et al*. 2011). Managing climate-driven range extension of these problematic species, for instance through physical removal, may help to alleviate the pressure on subtidal benthic invertebrates.

Johnson C.R., Banks S.C., Barrett N.S., Cazassus F., Dunstan P.K., Edgar G.J., Frusher S.D., Gardner C., Haddon M., Helidoniotis F. & Hill K. L. (2011) Climate change cascades: Shifts in oceanography, species' ranges and subtidal marine community dynamics in eastern Tasmania. *Journal of Experimental Marine Biology and Ecology*, 400, 17–32.
Kyle C., Cavanaugh J.R., Kellner A.J., Forde D.S., Gruner J.D., Parker W., Rodriguez I. & Feller C. (2014) Poleward expansion of mangroves is a threshold response to decreased frequency of extreme cold events. *Proceedings of the National Academy of Sciences*, 111, 723–727.
Pitt N.R., Poloczanska E.S. & Hobday A.J. (2010) Climate-driven range changes in Tasmanian intertidal fauna. *Marine and Freshwater Research*, 61, 963–970.

10.3. Transplant/release climate change-resistant captive-bred or hatchery-reared individuals to re-establish or boost native populations

- We found no studies that evaluated the effects of transplanting/releasing mate change-resistant captive-bred or hatchery-reared individuals on subtidal benthic invertebrate 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

Many marine species are vulnerable to climate change (Hoegh-Guldberg & Bruno 2010), but certain genetic strains or family lines within species can be more resistant and robust to the effects of climate change than others. Climate-resistant strains or families of certain species could be developed, for instance through selective breeding or genetic modification (van Oppen *et al.* 2015). For instance, selectively-bred family lines of the Sydney rock oyster *Saccostrea glomerata* have been shown to be more resilient to ocean acidification than the wild populations of Sydney rock oyster (not selectively-bred) (Parker *et al.* 2011). Transplanting or releasing climate change-resistant captive-bred or hatchery-reared species, for instance following the selection of a particular genetic strain or family-line, may potentially help re-establish or boost native populations by adding resistant individuals into the population and improving its genetic diversity (Bernhardt & Leslie 2013).

Evidence related to the transplantation of individuals outside of the context of climate change are summarised under "Species management – Transplant/release captive-bred or hatchery-reared species".

Bernhardt J.R. & Leslie H.M. (2013) Resilience to climate change in coastal marine ecosystems. *Annual Review of Marine Science,* 5, 371–392.

Hoegh-Guldberg O. & Bruno J.F. (2010) The impact of climate change on the world's marine ecosystems. *Science*, 328, 1523–1528.

Parker L.M., Ross P.M. & O'Connor W.A. (2011) Populations of the Sydney rock oyster, *Saccostrea glomerata*, vary in response to ocean acidification. *Marine Biology*, 158,3, 689–697.

van Oppen M.J., Oliver J.K., Putnam H.M. & Gates R.D. (2015) Building coral reef resilience through assisted evolution. *Proceedings of the National Academy of Sciences*, 112, 2307–2313.

10.4. Transplant captive-bred or hatchery-reared individuals of habitat-forming (biogenic) species that are resistant to climate change

- We found no studies that evaluated the effects of transplanting captive-bred or hatchery-reared individuals of habitat-forming/biogenic species that are resistant to climate change on subtidal benthic invertebrate 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

Many marine species are vulnerable to climate change (Hoegh-Guldberg & Bruno 2010), but preserving their habitats and promoting biodiversity is thought to help improve the overall resistance of the system and help alleviate the negative effects of climate change.

Transplanting individuals of habitat-forming (biogenic) species (such as kelp, seagrass, mussels, oysters or corals) resistant to climate change can potentially provide a climate-resistant habitat for associated species (van Oppen *et al.* 2015). These climate-resistant strains or families could be developed for instance through selective breeding or genetic modification (van Oppen *et al.* 2015). For instance, transplanting individuals of climate-resistant strains of reef-building oysters (Parker *et al*. 2011) grown in a hatchery may help create an oyster reef that will be resilient to climate change and beneficial to associated subtidal benthic invertebrates.

Evidence related to the transplantation of individuals outside of the context of climate change are summarised under "Habitat restoration and creation – Transplant captive-bred or hatchery-reared habitat-forming (biogenic) species".

Hoegh-Guldberg O. & Bruno J.F. (2010) The impact of climate change on the world's marine ecosystems. *Science*, 328, 1523–1528.
Parker L.M., Ross P.M. & O'Connor W.A. (2011) Populations of the Sydney rock oyster, *Saccostrea glomerata*, vary in response to ocean acidification. *Marine Biology*, 158,3, 689–697.
van Oppen M.J., Oliver J.K., Putnam H.M. & Gates R.D. (2015) Building coral reef resilience through assisted evolution. *Proceedings of the National Academy of Sciences*, 112, 2307–2313.

10.5. Limit, cease or prohibit the degradation and/or removal of carbon sequestering species and/or habitats

- We found no studies that evaluated the effects of limiting, ceasing or prohibiting the degradation and/or removal of carbon sequestering species and/or habitats on subtidal benthic invertebrate 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

Anthropogenic climate change is driven by atmospheric gases, including carbon dioxides (IPCC 2013). Certain marine species and habitats, such as oyster reefs, coral reefs, seagrass beds and macroalgae forests, can act as sinks for the carbon dioxide that gets absorbed in seawater (Dehon 2010; Mcleod *et al.* 2011). However, many of these carbon-sequestrating species are vulnerable to climate change (Hoegh-Guldberg & Bruno 2010). Limiting, ceasing or prohibiting the degradation and/or removal of these carbon sequestering species and habitats may potentially benefit subtidal benthic invertebrates through improved resilience and reduction in carbon dioxide levels.

Evidence for other interventions related to carbon sequestration are summarised under "Threat: Climate change and severe weather – Promote natural carbon sequestration species and/or habitats".

Dehon D.D. (2010) Investigating the use of bioengineered oyster reefs as a method of shoreline protection and carbon storage. Master's Thesis. Louisiana State University, 1084.
Hoegh-Guldberg O. & Bruno J.F. (2010) The impact of climate change on the world's marine ecosystems. *Science*, 328, 1523–1528.
IPCC (2013) *Climate Change 2013: The Physical Science Basis*. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex & P.M. Midgley (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.
Mcleod E., Chmura G.L., Bouillon S., Salm R., Björk M., Duarte C.M., Lovelock C.E., Schlesinger W.H. & Silliman B.R. (2011) A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO_2. *Frontiers in Ecology and the Environment,* 9, 552–560.

10.6. Promote natural carbon sequestration species and/or habitats

- We found no studies that evaluated the effects of promoting natural carbon sequestration species and/or habitats on subtidal benthic invertebrate 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

Anthropogenic climate change is driven by atmospheric gases, including carbon dioxides (IPCC 2013). Certain marine species and habitats, such as oyster reefs, coral reefs, seagrass beds and macroalgae forests, can act as sinks for the carbon dioxide that gets absorbed in seawater (Dehon 2010; Mcleod *et al.* 2011; Ware *et al.* 1992). Promoting the occurrence and persistence of these carbon sequestering species and habitats may potentially benefit subtidal benthic invertebrates through improved resilience and reduction in carbon dioxide levels.

Evidence for other interventions related to carbon sequestration are summarised under "Threat: Climate change and severe weather – Limit, cease or prohibit the degradation and/or removal of carbon sequestering species and/or habitats".

Dehon D.D. (2010) Investigating the use of bioengineered oyster reefs as a method of shoreline protection and carbon storage. Master's Thesis. Louisiana State University, 1084.

Hoegh-Guldberg O. & Bruno J.F. (2010) The impact of climate change on the world's marine ecosystems. *Science*, 328, 1523–1528.

IPCC (2013) *Climate Change 2013: The Physical Science Basis*. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex & P.M. Midgley (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Mcleod E., Chmura G.L., Bouillon S., Salm R., Björk M., Duarte C.M., Lovelock C.E., Schlesinger W.H. & Silliman B.R. (2011) A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO_2. *Frontiers in Ecology and the Environment,* 9, 552–560.

10.7. Create a Marine Protected Area or set levels of legal protection where natural climate refugia occur to further promote the persistence and recovery of species facing climate change

- We found no studies that evaluated the effects of creating a marine protected area or setting levels of legal protection where natural climate refugia occur to further promote the persistence and recovery of species facing climate change on subtidal benthic invertebrate 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

Anthropogenic climate change is driven by atmospheric gases, including carbon dioxides (IPCC 2013). Many marine species are vulnerable to climate change and likely to be impacted by it in the future (Hoegh-Guldberg & Bruno 2010). Climate refugia are areas naturally relatively buffered from contemporary climate change over time and that can enable the persistence of species and habitats (Keppel *et al*. 2015; Morelli *et al.* 2016). Marine protected areas restricting specific impactful activities, or other areas where legal protection or restriction occur, could be created where such refugia occur, to further enhance the refugia effect and promote the persistence and recovery of species facing climate change (Game *et al.* 2011; Green *et al.* 2014). Evidence for other interventions related to the creation of marine protected areas is summarised under “Habitat Protection”.

Hoegh-Guldberg O. & Bruno J.F. (2010) The impact of climate change on the world’s marine ecosystems. *Science*, 328, 1523–1528.

IPCC (2013) *Climate Change 2013: The Physical Science Basis*. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex & P.M. Midgley (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Game E.T., Lipsett-Moore G., Saxon E., Peterson N. & Sheppard S. (2011) Incorporating climate change adaptation into national conservation assessments. *Global Change Biology*, 17, 3150–3160.

Green A.L., Fernandes L., Almany G., Abesamis R., McLeod E., Aliño P.M., White A.T., Salm R., Tanzer J. & Pressey R.L. (2014) Designing marine reserves for fisheries management, biodiversity conservation, and climate change adaptation. *Coastal Management*, 42, 143–159.

Keppel G., Mokany K., Wardell-Johnson G.W., Phillips B.L., Welbergen J.A. & Reside A.E. (2015) The capacity of refugia for conservation planning under climate change. *Frontiers in Ecology and the Environment*, 13, 106–112.

Morelli T.L., Daly C., Dobrowski S.Z., Dulen D.M., Ebersole J.L., Jackson S.T., Lundquist J.D., Millar C.I., Maher S.P., Monahan W.B. & Nydick K.R. (2016) Managing climate change refugia for climate adaptation. *PLoS One*, 11, p.e0159909.

11. Habitat protection

Background

Habitat destruction is the largest single threat to biodiversity worldwide across systems, and habitat fragmentation and degradation often reduces the quality of remaining habitat (Brooks *et al.* 2002). Habitat protection is therefore one of the most frequently used conservation interventions both on land and in aquatic systems. Habitat protection can be achieved through the designation of legally protected areas, using international, national or local legislation (Davies *et al.* 2017), but also through voluntary designations (Prior 2011). Protection can be of entire habitat types, for example through the European Union's Habitats Directive, or occur on a smaller scale, restricting detrimental activities in a specific area (Zupan *et al*. 2018). It can be difficult to measure the effectiveness of legally protected areas as there may not be suitable controls and appropriate replication can be difficult to achieve (Fraschetti *et al*. 2002; 2005).

The past decades have seen a rise in the designation of protected areas in the marine environment, which restrict and manage human activities and disturbances (such as damaging fishing and mining practices), with the aim to protect important habitats both for biodiversity conservation and resource sustainability (Edgar *et al.* 2014; Kelleher & Kenchington 1991), or to protect representative samples of all habitat types with appropriate connectivity (Crochelet *et al.* 2016). This chapter describes interventions that can be used to benefit subtidal benthic invertebrate species by protecting the natural marine habitats they live in.

Note that evidence for similar interventions that are carried out outside marine protected areas can be found in other chapters, including "Threat: Biological resource use" (for fishing restrictions), "Threat: Pollution", "Threat: Human intrusions and disturbances" (for recreational activities restrictions), "Threat: Transportation and service corridors" (for shipping restrictions), and "Threat: Energy production and mining" (for restrictions on industrial activities such as aggregate extraction or mining).

Brooks T.M., Mittermeier R.A., Mittermeier C.G., Da Fonseca G.A., Rylands A.B., Konstant W.R., Flick P., Pilgrim J., Oldfield S., Magin G. & Hilton-Taylor C. (2002) Habitat loss and extinction in the hotspots of biodiversity. *Conservation Biology*, 16, 909–923.

Crochelet E., Roberts J., Lagabrielle E., Obura D., Petit M. & Chabanet P. (2016) A model-based assessment of reef larvae dispersal in the Western Indian Ocean reveals regional connectivity patterns—Potential implications for conservation policies. *Regional Studies in Marine Science*, 7, 159–167.

Davies T.E., Maxwell S.M., Kaschner K., Garilao C. & Ban N.C. (2017) Large marine protected areas represent biodiversity now and under climate change. *Scientific Reports*, 7, 9569.

Edgar G.J., Stuart-Smith R.D., Willis T.J., Kininmonth S., Baker S.C., Banks S., Barrett N.S., Becerro M.A., Bernard A.T., Berkhout J. & Buxton C.D. (2014) Global conservation outcomes depend on marine protected areas with five key features. *Nature*, 506, 216–220.

Fraschetti S., Terlizzi A., Bussotti S., Guarnieri G., D'Ambrosio P. & Boero F. (2005) Conservation of Mediterranean seascapes: analyses of existing protection schemes. *Marine Environmental Research*, 59, 309–332.

Fraschetti S., Terlizzi A., Micheli F., Benedetti-Cecchi L. & Boero F. (2002) Marine protected areas in the Mediterranean Sea: objectives, effectiveness and monitoring. *Marine Ecology*, 23, 190–200.

Kelleher, G., & Kenchington, R.A. (1991) Guidelines for establishing marine protected areas (Vol. 3). IUCN.

Prior S. (2011) *Investigating the use of voluntary marine management in the protection of UK marine biodiversity.* Report to Wales Environment Link.

Zupan M., Bulleri F., Evans J., Fraschetti S., Guidetti P., Garcia-Rubies A., Sostres M., Asnaghi V., Caro A., Deudero S. & Goñi R. (2018) How good is your marine protected area at curbing threats? *Biological Conservation*, 221, 237–245.

11.1. Designate a Particularly Sensitive Sea Area (PSSA) to regulate impactful maritime activities

- We found no studies that evaluated the effects of designating a Particularly Sensitive Sea Area on subtidal benthic invertebrate 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

Shipping, fishing, and other anthropogenic activities can impact subtidal benthic invertebrates through species removal or habitat damage. Specific areas of recognised scientific, ecological, or socio-economic significance can be designated as Particularly Sensitive Sea Area (PSSA) by the International Maritime Organization (IMO) because of their vulnerability and sensitivity to international maritime activities (IMO Resolution A.982(24); Kachel 2008; Lefebvre-Chalain 2007). Specific management measures can be taken within PSSAs to control for impactful maritime activities, such as ship routeing measures, the strict application of pollution legislation and waste or ballast discharge regulations, or the requirements for specific equipment on ships (Kachel 2008).

Inside PSSAs, the threat to subtidal benthic invertebrates from specific maritime activities is potentially removed or regulated, and previously impacted populations are, in theory, able to recover over time.

Related evidence linked with shipping regulations and interventions has been summarised under "Threat: Transportation and service corridors – Shipping lanes".

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

IMO Assembly Resolution 24/982 (2005) Revised guidelines for the identification and designation of Particularly Sensitive Sea Areas.

Kachel M.J. (2008) Particularly sensitive sea areas. Pages 1-184 in: *The IMO's Role in Protecting Vulnerable Marine Areas*. Berlin Heidelberg: Springer-Verlag.

Lefebvre-Chalain H. (2007) Fifteen years of particularly sensitive sea areas: a concept in development. *Ocean & Coastal Law Journal*, 13, 47.

11.2. Designate a Marine Protected Area and prohibit all types of fishing

- **Thirty studies** examined the effects of prohibiting all types of fishing in marine protected areas on subtidal benthic invertebrate populations. Four studies were systematic reviews of marine reserves (New Zealand[13] and across the world[16,24,26]). Two studies were in the North Atlantic Ocean[1,5] (Bahamas). Five were in the South Pacific Ocean[2,3,8,12,23] (New Zealand, French Polynesia). Three were in the North Pacific Ocean[4,7,10] (USA). Seven were in the Tasman Sea[6,14,15,17a–b,20,22] (New Zealand, Australia). One was in the Florida Keys[9] (USA). One was in the Coral Sea[11] (Australia). Three were in the Mediterranean Sea[18,21,27] (Italy, Spain). One was in the Bristol Channel and the Irish Sea[19] (UK). Two were in the Firth of Clyde[25,29] (UK). One was in the Foveaux Straight[28] (New Zealand).

COMMUNITY RESPONSE (5 STUDIES)

- **Overall community composition (3 studies):** Three site comparison studies (one replicated and paired, one replicated, one paired) in the Mediterranean Sea[21], the Tasman Sea[22], and the Firth of Clyde[25] found that marine protected areas that had been prohibiting all fishing for five to 16 years depending on the study, had similar combined algae, invertebrate and fish community composition[21], similar combined mollusc and echinoderm community composition[22], and similar overall community composition of large invertebrates[25] but different composition of small sessile invertebrates[25], compared to fished areas.

- **Overall species richness/diversity (5 studies):** One global systematic review[16], and three site comparison studies (one replicated and paired, one replicated, one paired) in the Mediterranean Sea[21], the Tasman Sea[22], and the Firth of Clyde[25] found that marine protected areas that had been prohibiting all fishing for five to 16 years depending on the study, had similar overall invertebrate species richness/diversity[16,25], similar combined algae, invertebrate and fish species richness[21], and similar combined mollusc and echinoderm species richness[22], compared to fished areas. One site comparison study in the Tasman Sea[14] found inside a marine protected area prohibiting all mobile fishing that macroinvertebrate species richness remained stable over the 15 years after its designation and enforcement, but decreased at fished sites.

POPULATION RESPONSE (2 STUDIES)

- **Overall abundance (4 studies):** Two systematic reviews of marine protected areas across the world prohibiting all fishing[16,26] found that they had greater overall invertebrate abundance and biomass compared to fished areas. Two site comparison studies (one before-and-after, one replicated) in the Tasman Sea[14,22] found that inside marine protected areas prohibiting all fishing, overall invertebrate abundance did not change over the 15 years after their designation and enforcement and that it did not change in fished areas either[14], and that all areas had similar combined mollusc and echinoderm abundance after 16 years[22].
- **Overall condition (1 study):** One global systematic review[16] found that in marine protected areas prohibiting all fishing, invertebrates were bigger compared to fished areas.
- **Crustacean abundance (17 studies):** Two reviews (one global and systematic, one of New Zealand areas) found that marine protected areas prohibiting all fishing had more lobsters compared to marine protected areas only partially prohibiting fishing[24] and unrestricted fished areas[13]. Nine of 15 site comparison studies (including replicated, randomized, paired, before-and-after) in the North Atlantic Ocean[5], the Bristol Channel and the Irish Sea[19], the Firth of Clyde[29], the Mediterranean Sea[18,27], the North Pacific Ocean[10] the Florida Keys[9] the South Pacific Ocean[3], the Tasman Sea[6,14,15,17a,20], and the Coral Sea[11] found that inside marine protected areas prohibiting all fishing, the abundances and/or biomasses of lobsters[3,9,12,14,17a,18,20,27] and mud crabs[11] were higher compared to areas where seasonal or unrestricted fishing was allowed, after four to 33 years depending on the study. Four found that they had mixed effects on the abundances of lobster[5,15,] and crab species[19,29], after one to seven years depending on the study. Two found that they had similar abundance of lobsters compared to fished areas after either five to seven years[6] or after approximately 30 years[10].
- **Crustacean reproductive success (4 studies):** Two site comparison studies (one replicated, randomized) in the Florida Keys[9] and the Firth of Clyde[29] found that marine protected areas prohibiting all fishing and harvesting had similar population sex ratios of lobsters compared to where seasonal fishing[9] or all fishing[29] was allowed, after four to seven years depending on the study. Two replicated, site comparison studies (one randomized) in the Tasman Sea[17b] and the Mediterranean Sea[27] found that marine protected areas prohibiting all fishing had greater lobster egg production potential compared to commercial fishing exclusion zones[17b] and fully fished areas[27], after either 15 years[17b] or 21 to 25 years[29]. One site comparison study in the Firth of Clyde[29] found that marine protected areas prohibiting all fishing had more female lobsters with eggs than fished areas, after four to seven years.
- **Crustacean condition (8 studies):** One review of studies in New Zealand[13], and five of seven site comparison studies (four replicated, one replicated and randomized) in the North Atlantic Ocean[5], the Bristol Channel and the Irish Sea[19], the Firth of Clyde[29], the Florida Keys[9], the South Pacific Ocean[3], the Coral Sea[11], and the Tasman Sea[6], found that marine protected areas prohibiting all fishing had bigger lobsters[12,6,9,3,29] and crabs[11] compared to seasonally fished or

fully fished areas, after four to seven years depending on the study. Three found mixed effects on lobsters[5,19] and crabs[19,29] depending on species[19,29], sex[5], and locations[5], after one to seven years depending on the study.

- **Crustacean population structure (2 studies):** Two replicated site comparison studies (one randomized) in the Tasman Sea[17b] and the Mediterranean Sea[27] found that marine protected areas prohibiting all fishing had different population size structures of lobsters compared to commercial fishing exclusion zones (only for females)[17b] and compared to fished areas[27], after either 15 years[17b] or 21 to 25 years[27].
- **Echinoderm abundance (3 studies):** Two of three site comparison studies (two replicated, one paired) in the North Pacific Ocean[10], the South Pacific Ocean[2], and the North Pacific Ocean[4], found that marine protected areas prohibiting all fishing had similar abundance of Kina sea urchins after more than 10 years[2], and sea cucumbers after eight years[4] to fished areas, and one found higher abundance of red sea urchins after approximately 30 years[10]. One also found that the effects on abundance of red sea urchins depended on the age of the protected area and the size of the urchins[4].
- **Echinoderm condition (1 study):** One paired, site comparison study in the South Pacific Ocean[2] found that marine protected areas that had been prohibiting all fishing for over 10 years had heavier Kina sea urchins compared to fished areas.
- **Mollusc abundance (10 studies):** Four of 10 site comparison studies (including replicated before-and-after, and site comparison) in the North Atlantic Ocean[1], the North Pacific Ocean[4,7,10] the South Pacific Ocean[23,8], the Tasman Sea[14,15,22], and the Foveaux Straight[28] found that inside a marine reserve prohibiting all fishing, abundances/biomass of giant clams[23], adult queen conch[1], Cook's turban snails[8], rock scallops[10] and green abalone[10] were higher compared to a fished area, after eight to 36 years depending on the study. Six found similar abundances of scallop species[4,28], pink abalone[7,10], juvenile queen conch[1], and top shell species[8], after five to 36 years depending on the study. Three found lower abundances of star limpets after 23 to 25 years[8] and blacklip abalone after 15 to 16 years[14,22]. One found that the effects of marine protected areas prohibiting all fishing on the abundance of mussel species compared to a commercial fishing exclusion zone varied with the age and location of the protected areas[14].
- **Mollusc reproductive success (1 study):** One site comparison study in the North Atlantic Ocean[1] found that inside a marine protected area that had been prohibiting all fishing for 33 to 36 years, abundance of queen conch larvae was higher compared to an unprotected fished area.
- **Mollusc condition (1 study):** One site comparison study in the North Pacific Ocean[7] found that in marine protected areas that had been prohibiting all fishing pink abalone were bigger five to 23 years after their designation, compared to fished site.

Background

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear entering in contact with the seabed (Collie *et al*. 2000). Specific areas can be designated as protected, and specific management measures taken to control for impactful activities, such as prohibiting all types of fishing (Villamor & Becerro 2012). Such areas are often referred to as marine reserves, marine sanctuaries, or more commonly "no-take areas/zones". Inside such protected no-take areas, the threat to subtidal benthic invertebrates is removed, and previously impacted populations are, in theory, able to recover over time (Jack & Wing 2010).

When this intervention occurs outside of a protected area, evidence has been summarised under "Threat: Biological resource use – Cease or prohibit all types of fishing".

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.
Jack L. & Wing S.R. (2010) Maintenance of old-growth size structure and fecundity of the red rock lobster *Jasus edwardsii* among marine protected areas in Fiordland, New Zealand. *Marine Ecology Progress Series,* 404, 161-172.
Villamor A. & Becerro M.A. (2012) Species, trophic, and functional diversity in marine protected and non-protected areas. *Journal of Sea Research*, 73, 109–116.

A site comparison study in 1988–1994 in two sandy areas with seagrass in Exuma Cays, North Atlantic Ocean, central Bahamas (1) found that inside a protected marine reserve closed to all fishing for 33–36 years, abundance of adult and larval queen conch *Strombus gigas* were higher, but abundance of juveniles was similar, compared to a nearby unprotected fished area. Adult conch abundance was higher in the closed area (34–270 conch/ha), compared to the fished area (2–88 conch/ha). In six of 13 comparisons, larval abundance was significantly higher in the closed area (1–50 larvae/10 m^3) compared to the fished area (0.06–6), and statistically similar in seven comparisons (closed: 0.2–55; fished: 0.25–1.6 larvae/10 m^3). Juvenile conch abundance was statistically similar in closed (2–6 conch/ha) and fished areas (0–2). The marine reserve (456 km^2) was designated in 1958, closed to all fishing and the collection of any animals prohibited. Inside the fished area, taking juvenile queen conch and using SCUBA gear for fishing is prohibited. A snorkeller counted adult queen conch along 12 transects (6 m wide; total area of 28 ha) in March–September 1991 (fished area) and 1994 (closed area). Divers also measured adult conch abundance, shell length and lip thickness (see paper for details). Juvenile conch abundance was estimated in each area (annually in 1988–1991 in the fished area; in 1991 in the closed area; see paper for details). Queen conch larvae were surveyed in June–August 1993–1994 using plankton nets.

A paired, site comparison study in 1992–1997 of six coralline seabed areas in the South Pacific Ocean, northeastern New Zealand (2) found that inside marine protected areas prohibiting all fishing for more than 10 years Kina sea urchins *Evechinus chloroticus* tended to be heavier compared to adjacent fished areas, but that all areas appeared to have similar urchin abundance. Results were not statistically tested. Sea urchins tended to be heavier inside protected areas (average 54 g), compared to fished areas (40 g). Sea urchin abundance varied between 1 to 4 urchin/m^2 in protected areas, and 1 to 7 urchin/m^2 in fished areas. Three marine protected areas prohibiting all exploitation (assumed to include all fishing) and three paired adjacent fished areas were sampled between 1992 and 1997 (4–5 sites/area; 5–10 m depth). Divers counted all urchins in twenty 1 m^2 quadrats and measured the diameter of at least 60 urchins. In one protected and one fished area, the skeletons of ten sea urchins (55–65 mm diameter)/site were wet-weighed.

A replicated, paired, site comparison study in 1995 in eight sites of kelp beds around islands in the South Pacific Ocean, northeastern New Zealand (3) found that marine protected areas prohibiting all fishing (no-take) for 20 years had more and bigger spiny lobsters *Jasus edwardsii* compared to adjacent fished areas. Abundance of lobsters was greater in no-take (455 lobsters/ha) compared to fished areas (174 lobsters/ha). In addition, average lobster size was greater in no-take (110 mm), compared to fished areas (94 mm). Leigh Marine Reserve (established 1975) was surveyed in spring 1995, and Tāwharanui Marine Park (established 1982) in autumn 1995. At two sites within and two sites outside each no-take area, divers counted and visually estimated the carapace length of all lobsters within five randomly-placed 50 × 10 m transects (max. 25 m depth).

A replicated, site comparison study in 1998 of eight rocky and sandy areas in the San Juan Archipelago, northwest Pacific Ocean, USA (4) found that the effects of prohibiting all fishing and harvesting within marine protected areas (no-take) on the abundance of red sea urchins *Strongylocentrotus franciscanus* depended on the age of the protected area and the size of the urchins, and that no-take areas did not affect abundances of sea cucumbers *Parastichopus californicus* and scallops *Chlamys rubida*, *Chlamys behringiana*, and *Hinnites giganteus*. Abundances of medium and large urchins were higher in the eight-year-old no-take areas (medium: 65; large: 225 individuals/300 m^2) compared to one-year-old no-take areas (medium: 21; large: 43) and unprotected areas (medium: 9; large: 21). There were no significant differences between young protected areas and unprotected areas. Abundance of small urchins was similar across areas (1–4). Abundance data for sea cucumbers and scallops were not provided. The authors suggest the lack of increase in abundances inside protected areas was likely due to a lack of compliance and enforcement of prohibitions. In July 1998, three marine preserves (established eight years prior and prohibiting the harvest of organisms; sea urchin fishery closed since the late 1970s), two marine protected area (designated in 1997; voluntary no-take zones), and three unprotected areas were surveyed. Divers counted and measured red sea urchins, sea cucumbers, and scallops along 300 m^2 transects (4 transects/area).

A site comparison study in 1994–1995 in four coral reef areas in Exuma Sound, North Atlantic Ocean, Bahamas (5) found that the effects of prohibiting all fishing within marine protected areas (no-take) on the abundance and size of Caribbean spiny lobsters *Panulirus argus* after 8–9 years varied between areas, year and seasons. Overall, lobster abundance was higher in the no-take area (63–51 lobsters/ha) than in only one of three fished areas (data not shown), and similar to abundance in the other two fished areas (6–333 lobsters/ha). The closed area had larger males (118 mm) than two (105–106 mm) of three fished areas (no difference to the other; 113 mm). However, the closed area had larger females (109 mm) than only one (95 mm) of three fished areas (no difference to the other two; 108–128 mm). In 1986, a 456 km^2 marine reserve (established in 1959) was designated as a no-take area (although some poaching still occurred). In spring (before the fishing season) and autumn (after the fishing season) 1994–1995, lobster abundance was recorded in the reserve and in three adjacent fished areas (3–17 reef sites/area). Divers examined crevices at depths <20 m, and measured (carapace length), and sexed all lobsters found.

A site comparison study in 1994–2000 of twelve rocky seabed and soft sediment sites in Tonga Island Marine Reserve, Tasman Sea, New Zealand (6) found that five to seven years after designating a marine protected area prohibiting all fishing (no-take reserve), abundance of spiny lobsters *Jasus edwardsii* was similar inside the protected area, compared to outside where fishing occurred, but lobster size was higher. Average abundance was not statistically higher inside the reserve (1.3 lobsters/100 m^2) compared to outside (0.4 lobsters/100 m^2). Lobster were larger inside the reserve 113–132 mm), compared to outside (94–104 mm; results not statistically tested). In addition, when compared with data from 1994 (12 months after establishing the reserve) average lobster abundance increased inside the reserve (1994: 1; 1998–2000: 1.4 lobster/100 m^2) but decreased outside (1994: 0.7; 1998–2000: 0.6 lobster/100 m^2; results not statistically tested). Tonga Island Marine Reserve was established in 1993 and is closed to all fishing and harvesting. Seven sites inside and five sites outside the reserve (6–11 m depth) were surveyed on six occasions in 1998–2000. During each survey, divers counted and estimated the size of lobsters in twelve 24 x 4 m transects/site. Abundance data from 1998 and 2000 (obtained in December) were compared with a prior survey in December 1994 of five sites inside and five sites outside the reserve (10–12 transects/site; 30 × 4 m).

A site comparison study in 1983–2001 of three sites of kelp forest in the Channel Islands National Park, southern California, North Pacific Ocean, USA (7) found that the effects of designating protected areas prohibiting all fishing (no-take) on abundance, size, and egg production of pink abalone *Haliotis corrugata* depended on the level of enforcement. Five to 23 years after their designation, cumulative abundance across years was higher in an enforced (333 abalone/13,040 m) compared to a not enforced (116) no-take area, but these were not significantly different from a fished site (431). Abundance declined over time at all sites. Size of abalones was higher in the enforced no-take area (147 mm) and in the unenforced no-take area (134 mm) compared to the fished site (122 mm). More large abalone (above minimum landing size of 158 mm) were found in the enforced no-take area (30%) and in the unenforced no-take area (6%) compared to the fished site (2%). Egg production was higher in the enforced no-take area (2,555; units unclear) compared to the other sites (unenforced no-take: 550; fished site: 1,420). Annually between 1983 and 2001, pink abalone were counted and measured by divers along 10–12 transects (40–60 m^2) at three sites. Two were no-take areas established in 1978 (one enforced, one not enforced) and one a site where non-commercial fishing occurred (commercial fishing was prohibited). Egg production was estimated from abundance and size data.

A site comparison study in 1999–2001 of eight sites in two rocky areas in the South Pacific Ocean, New Zealand (8) found that designating a marine protected area prohibiting all fishing (no-take reserve) had mixed effects on invertebrate abundances depending on species, after 23–25 years. Abundance of Cook's turban snails *Cookia sulcata* was higher at sites inside the reserve (2–16 individuals/m^2) compared to fished sites outside (1–2), but abundance of star limpets *Cellana stellifera* was lower inside (0–2). compared to outside (0–6). Abundances were similar at sites inside and outside the reserve for the green top shells *Trochus viridis* (inside: 0–12; outside: 0–10) and red opal top shells *Cantharidus purpureus* (inside: 1–14; outside: 0–22). Leigh Marine Reserve (no-take area) was established in 1975 (date taken from (3) summarised above). Annually in summer in 1999–2001, four sites inside and four outside the reserve were sampled. Invertebrates were counted in twenty 1 m^2 quadrats/site (2–10 m depth).

A replicated, randomized, site comparison study in 1997–2001 in two areas of rocky reef in the Florida Keys, USA (9) found that inside a protected marine reserve prohibiting all fishing and harvesting, the size and abundance of Caribbean spiny lobsters *Panulirus argus* were higher compared to outside the reserve where seasonal fishing was allowed, but the population sex ratio was similar. On average over the four years following its designation, lobsters were bigger inside the reserve (82–94 mm carapace length), compared to outside (77–85 mm). There were more lobsters bigger than 76.2 mm (legal-catch size) in the reserve (3–86/60 min search) compared to outside (0.7–71/60 min search). The population sex ratio was not significantly different inside (0.5–3 female:male) and outside the reserve (0.4–5). In addition, inside the reserve, average lobster sizes increased during the study (1997: 82–86; 2001: 87–94 mm). In July 1997, a 3,000-ha reserve was established prohibiting all fishing and harvesting year-round. Annually in July and September of 1997–2001, divers counted, measured, and recorded the sex of lobsters during twenty-four 60 min timed-surveys at randomly-chosen locations; twelve inside and twelve outside the reserve.

A replicated, site comparison study in 2002–2003 in one area of mixed seabed off the coast of southern California, North Pacific Ocean, USA (10) found that a protected marine reserve prohibiting all fishing had higher abundances of three out of five surveyed invertebrate species compared to outside after approximately 30 years. Average abundances were higher inside the reserve than outside the reserve for red sea urchins *Strongylocentrotus franciscanus* (inside: 0.86/m^2 vs outside: 0.02/m^2), rock scallops *Crassadoma giganteum* (0.006/m^2 vs 0.001/m^2), and green abalone *Haliotis fulgens* (0.007/m^2 vs 0.001/m^2), but not for California spiny lobsters *Panulirus interruptus* (0.006 vs 0.004/m^2) and pink abalone *Haliotis corrugata* (0.003/m^2 vs 0.004/m^2). A 2.16 km^2 area was established as a protected no-take marine reserve in 1971. Between spring and summer 2002, a total of 286 transects (30 × 4 m) were surveyed by divers inside and outside of the reserve (numbers of transects unspecified), and the abundances of red sea urchins, rock scallops, spiny lobsters, and pink abalone recorded. Green abalone abundance was recorded during dive surveys along 500 m transects (numbers unspecified) at depths below 6 m between spring and autumn 2003.

A replicated, paired, site comparison study in 2002–2003 of six areas of muddy seabed and seagrass in Moreton Bay, Coral Sea, Australia (11) found that marine protected areas prohibiting all fishing (no-take reserves) had typically more and bigger mud crabs *Scylla serrata*, compared to nearby and distant fished areas, five years after designation. Abundance of mud crabs was approximately 2.5 times higher in the reserves (1.5 crabs/pot) compared to three of four fished areas (0.3–0.5 crabs/pot), but not significantly different from one nearby fished area (1.1 crabs/pot). Mud crabs were on average bigger in the reserves (15.7–16.1 cm), compared to the fished areas (14.7–15.8 cm). Two no-take marine reserves (closed to all fishing) were established in 1997. Mud crabs were surveyed in the reserves and in four fished (recreationally and/or commercially, see paper for details) non-reserves (two paired with each reserve; one nearby ≤7 km away, one distant (distance unspecified)) in summer and winter 2002–2003. During each survey, 11 baited crab pots/area were deployed at 1–4 m depth (≥50 m apart), for 24 h and recovered (repeated two consecutive days). Upon recovery, all crabs captured were identified, measured (carapace width), and released. The relative abundance of crabs was expressed as catch/unit effort (meaning the number of crabs caught/pot).

A replicated, before-and-after, site comparison study in 1977–2005 of 10 rocky sites in the South Pacific Ocean, northeastern New Zealand (12) found that during the 22 years after implementing a protected marine park prohibiting all fishing (no-take), abundance and biomass of spiny rock lobsters *Jasus edwardsii* increased and became greater than at adjacent fished sites. Before designation, lobster abundance and biomass were similar inside (4–19 lobsters/transect; 0–3 kg/transect) and outside (0–7 lobsters/transect; 0.1 kg/transect) the park. Legal-size lobsters (>95 mm carapace length) in the park were

10.9 times more abundant after implementation (6–32 lobsters/transect) compared to before (0–5 lobsters/transect), with biomass 25 times higher (before: 0–3; after: 14–41 kg/transect). There was no change in abundance of sublegal-size lobsters inside the park. No legal-size and only 0–7 sublegal-size lobsters/transect were present outside the park after implementation. Tāwharanui Marine Park was established in 1981 (implemented 1983). Between 1977 and 2005, five no-take sites and five fully-fished sites outside the park were surveyed annually. Divers counted all lobsters and visually estimated the size and weight of legal-size lobsters along a 50 × 10 m fixed transect at each site.

A review of 14 studies undertaken between 1985 and 2002 in 20 areas of seabed in New Zealand (13) found that marine reserves prohibiting all fishing (no-take) typically had bigger and more abundant spiny rock lobsters *Jasus edwardsii* compared to fished areas outside the reserves. In 12 of 13 studies, rock lobsters were bigger inside the reserves (98 mm) than outside (79 mm), and in 11 of 14 studies lobster were more abundant inside the reserves (0.03 lobsters/m^2) than outside (0.01 lobsters/m^2). Older and larger reserves had greater effects than younger and smaller ones on lobster size (data presented as effect sizes). Size and abundance data were extracted from 14 studies of 10 marine reserves and 10 corresponding fished areas and used in a meta-analysis. At the time of surveys, the reserves were on average 8.5 years old.

A before-and-after, site comparison study in 1992–2007 in twelves rocky sites in the Tasman Sea, Australia (14) found that over the 15 years after designating and enforcing a marine protected area prohibiting all fishing, mobile macroinvertebrate species richness remained stable at protected sites but decreased at fished sites, while overall abundance did not change at any sites. Before enforcement, total macroinvertebrate species richness was lower at protected sites (11 species) compared to fished sites (16). After 15 years, species richness remained stable within protected sites (10–12) but had decreased in fished sites to similar levels (13–14). Before enforcement, overall mobile macroinvertebrate abundance was lower at protected sites (330–560 individuals/site) than fished sites (760–1,030) and remained similar at all sites over 15 years (protected: 375–430; fished: 625–820). This pattern was due to opposing changes in abundances of specific groups and species (see paper for details). In addition, abundance of blacklip abalone *Haliotis rubra* decreased over time inside the protected sites relative to fished sites, while abundance of southern rock lobsters *Jasus edwardsii* increased in protected sites but decreased in fished sites (data not provided). An area within Maria Island National Park was declared a no-take area in 1991 and closed to all fishing. In spring 2006 and autumn 2007, a diver visually identified and counted all mobile macroinvertebrates (echinoderms, crustaceans, and molluscs >1 cm) along four 50 m transects at six sites inside and six outside the no-take area (5 m water depth). Data were compared to historical surveys in spring and autumn 1992 before effective enforcement.

A replicated, site comparison study in 2006–2007 of 11 rocky seabed sites in Fiordland, Tasman Sea, New Zealand (15) found that the effects of marine protected areas prohibiting all fishing (no-take reserves) on the abundance of red rock lobsters *Jasus edwardssi* and percentage cover of its prey, the mussels *Mytilus edulis galloprovincialis*, *Perna canaliculus*, and *Aulacomya maoriana*, compared to a commercial fishing exclusion zone, varied with the age and location of the protected areas. Lobster abundance was higher in the >13-year-old no-take reserve established in 1993 (36 individuals/250 m^2) compared to one of two <2-year-old no-take reserves established in 2005 (Kutu Parera;

11 individuals/250 m^2) and the commercial exclusion zone (9 individuals/250 m^2). The second <2-year-old no-take reserve (Taipari Roa) had no lobsters. Mussel cover was higher in the >13-year-old reserve (28%) compared to the exclusion zone (15%), and cover at Kutu Parera (18%) was not different from either the >13-year-old reserve or the exclusion zone. No mussels were found at Taipari Roa. Lobsters and mussels were surveyed by divers at two sites in each of the following: a no-take reserve established in 1993, and two no-take reserves established in 2005, and at five sites within a commercial exclusion zone set in 2005 (15 m depth). During six surveys in 2006–2007, red rock lobsters were counted along 50 × 10 m transects (1–4 transects/site/survey). The percentage cover of mussels (species combined) was estimated from 25 photographs (0.17 m^2)/site taken during a single survey in 2007.

A systematic review of 149 studies published between 1977 and 2006 of no-take marine reserves across the world (16) found that inside marine protected areas prohibiting all fishing, invertebrate biomass, abundance, and size were greater, but species richness was not, compared to unprotected areas outside. Inside the reserves, average biomass increased by 752%, average abundance by 176%, and average size by 26%, compared to outside the reserve. Species richness decreased by a non-significant <5% inside compared to outside the reserves. When analysed by species group, molluscs and arthropods had the greatest increases (molluscs: +240% (non-significant) biomass, +422% abundance, +33% size; arthropods: +889% biomass, +323% abundance, +33% size), while there were no significant changes for any metrics for echinoderms or cnidaria. Species highly targeted by fisheries had the greatest increases in abundance (+385%) and biomass (+820%) in the reserves. The selected studies compared invertebrate abundance, biomass, size, and species richness for 124 reserves across 29 countries. Selected studies included comparisons of before-and-after the reserves were established, and comparisons of inside vs outside the reserves. A meta-analysis was performed on the selected studies.

A replicated, site comparison study in 2006–2007 of 26 rocky seabed sites in Fiordland, Tasman Sea, New Zealand (17a) found that older marine protected areas prohibiting all fishing (no-take reserves) had more red rock lobsters *Jasus edwardsii* compared to younger ones, to a commercial fishing exclusion zone and to adjacent areas without designated protection. Lobster abundance was higher in no-take reserves >13-year-old established in 1993 (12 individuals/250 m^2) compared to those <2-year-old established in 2005, commercial exclusion zones, and adjacent unprotected areas which had similar abundances to each other (1–2 individuals/250 m^2). In 2006 and 2007, divers surveyed four no-take reserves established in 1993, ten no-take reserves established in 2005, eight sites within a commercial fishing exclusion zone set in 2005, and four unprotected fished sites. All sites were located at 15 m depth on rocky habitat. Red rock lobsters were counted along 50 × 5 m transects (1 transect/site in 2006, 3/site in 2007).

A replicated, site comparison study in 2008 of eight rocky seabed sites in Fiordland, Tasman Sea, New Zealand (17b) found that marine protected areas prohibiting all fishing (no-take reserves) had different population structures of female red rock lobsters *Jasus edwardsii*, but not males, and greater egg production potential, compared to commercial fishing exclusion zones, but the effects varied with the age of the reserves. Population structure data were reported as size-frequency distributions. A 15-year-old reserve had greater abundance and size of female lobsters compared to commercial exclusion zones.

One of two 3-year-old reserves had no lobsters (either male or female); in the other abundance and size of female lobsters were not significantly different to the other sites. Egg production was higher in the 15-year-old reserve (8,350/m^2/year) compared to the commercial exclusion zones (1,260/m^2/year). The 3-year-old reserve with lobsters had an egg production not significantly different to the other sites (3,400/m^2/year). In 2008, divers surveyed two sites in each of the following: a no-take reserve established in 1993, two no-take reserves established in 2005, and a commercial fishing exclusion zone set in 2005. All sites were located at 15 m depth on rocky habitat. Red rock lobsters were counted along three 50 × 5 m transects, and their size and sex assessed from video footage (see study for details). Egg production potential was estimated using abundance and size data for female lobsters.

A before-and-after, site comparison study in 1997–2009 in areas of soft seabed in the western Mediterranean Sea, off the coast of Sardinia, Italy (18) found that, over the 10 years after designating a marine protected area prohibiting all fishing (no-take), abundance and biomass of European spiny lobster *Palinurus elephas* increased inside the no-take area, and after 10 years was greater than in an adjacent fished area. Within the no-take area lobster abundance increased over time (1998: 0.4 lobsters/net; 2009: 1.5 lobsters/net) and so did biomass (1997: 0.09 kg/net; 2009: 0.56 kg/net). In 2008–2009 (after 10 years) lobster abundance within the no-take area was 4.7 times greater than in the fished area (data not shown). The Su Pallosu no-take area was closed to fishing in 1998. Annually from 1997 to 2009, trammel nets (50 m long) were towed inside the no-take zone (91 nets in total; 2–14 nets/year). Similar nets were towed in an adjacent fished area in 2008 and 2009 (256 nets; within 25 km^2 of the no-take zone). Lobsters in each net were counted and weighed, and total abundance and biomass for each area estimated.

A replicated, site comparison study in summer 2004–2007 of six sites in three rocky and sandy seabed areas in the Bristol Channel and the Irish Sea, UK (19) found that a marine protected area prohibiting all fishing (no-take) had mixed effects on the abundances and sizes of European lobster *Homarus gammarus,* velvet crab *Necora puber,* brown crab *Cancer pagurus* and spider crab *Maja squinado*. Abundance of large lobsters (≥90 mm) increased by 127% inside the no-take zone between 2004 and 2007 (from 3 to 7 lobsters/line) and was five times higher than in unprotected fished areas where abundance had not changed (1–2 lobsters/line). Abundance of small lobsters (<90 mm) increased by 97% (from 3 to 7 lobsters/line) in the no-take zone, but remained constant in the fished areas (2–4 lobsters/line). The size of large lobsters (≥90 mm) increased by 5% inside the no-take zone between 2004 (98 mm) and 2007 (103 mm) and became 9% larger than in the fished areas where lobster size decreased by 2% (from 98 to 95 mm). The size of small lobsters did not change over time and was similar across all areas. Abundance of velvet crabs decreased by 65% inside the no-take zone over time (from 2 to 1 crabs/line; likely due to increased predation by lobsters) but increased in the fished areas (from 0–6 to 1–7 crabs/line). The average size of velvet crabs did not change over time and was similar across all areas. Abundance of brown crabs did not change over time inside the no-take zone (0.3 crab/line), nor in the fished areas (from 0.3–2 crabs/line). The average size of brown crabs increased by 25% inside the no-take zone between 2004 (115 mm) and 2007 (144 mm) but was not greater than in fished areas (116–130 mm). Abundance of spider crabs was similar in 2004 and 2007 for all areas but varied spatially (with the no-take zone having a lower abundance). The average size of spider crabs did not change over time inside the no-take zone. Lundy Island Marine Protected Area was designated as a voluntary reserve in 1971 (statutory since 1986). In 2003, it included a 4 km^2 no-take zone (no fishing or harvesting allowed), the rest being a refuge zone only allowing crab and lobster potting. In 2004–2007, lobsters and crabs were surveyed inside the no-take zone and two unprotected fished locations (20–100 km away) (2 sites/location). Four lines of standard commercial baited shellfish pots were deployed (10 pots/line) at each site for 24 h. Upon retrieval, lobsters and crabs were counted and measured (carapace length). The pots were redeployed for five consecutive days each year.

A replicated, site comparison study in 2004 in two rocky reef areas in the Tasman Sea, Australia (20) found that marine protected areas prohibiting all fishing (no-take reserves) had more spiny rock lobsters *Jasus edwardsii* compared to fished areas outside the reserves. After 12 and 33 years, average lobster abundances were higher inside the two reserves (12 years: 0.8; 33 years: 1.7 lobsters/100 m^2), compared to outside where lobsters were absent (0 lobster/100 m^2). Maria Island Marine Reserve was established in 1992. Crayfish Point Marine Reserve was established in 1971. Abundance of lobsters bigger than 140 mm carapace length was recorded along six 50 × 4 m transects during 15 min standardized timed-searches (six inside and six outside each reserve).

A replicated, paired, site comparison study in 2008 of ten rocky seabed areas in the Mediterranean Sea, Spain (21) found that marine protected areas prohibiting all fishing (no-take) for at least 10 years had similar overall combined invertebrate, fish, and algae community composition and species diversity to unprotected fished areas. Overall community composition varied geographically, but not with protection status (data presented as graphical analyses). Species diversity was similar across areas (diversity presented as a diversity index). In addition, two of ten filter-feeding invertebrate groups were more abundant inside the no-take areas than outside (see paper for details). In August 2008, three sites inside each of five no-take areas (designated between 1983 and 1998) and three sites in each of five adjacent (paired) fished areas were surveyed at 4–11 m depth. All organisms (invertebrates, fish, and algae), were identified and counted along a 50 m^2 transect/site. Filter-feeding invertebrates were sorted into 10 trophic groups.

A replicated, site comparison study in 2006–2007 of twelve rocky reef sites in the Tasman Sea, Australia (22) found that sites within a marine reserve prohibiting all fishing for 16 years had statistically similar combined mollusc and echinoderm species richness, abundance, and community composition to sites outside the reserve subject to fishing. Species richness and abundance data were not reported. Community composition data were reported as a graphical analysis. Abundance data were reported for the commercially valuable but declining blacklip abalone *Haliotis rubra* and was lower inside (0.25–0.29 abalone/boulder) compared to outside (0.62–0.94;) the reserve. In summer 2006–2007, ten boulders (30 × 30 × 5 cm) were deployed at each of 12 sites: six within the reserve (declared in 1991) and six outside. Boulders were recovered in January 2007 after three months, and molluscs (including blacklip abalone) and echinoderms present on their underside identified and counted. This was repeated in autumn and winter 2007.

A before-and-after, site comparison study in 2004 and 2012 in 18–19 sites of sandy and coral seabed in Tatakoto Atoll, South Pacific Ocean, French Polynesia (23) found that over the eight years after designating a marine protected area prohibiting all fishing (no-take), abundance and biomass of giant clams *Tridacna maxima* appeared higher inside compared to outside the no-take area, but were decreasing in both areas. Results were not tested for statistical significance. Abundance of clams decreased inside (from 119 to 11 clams/m^2) and outside (from 9 to 3 clams/m^2) the no-take zone. Abundances in 2012 corresponded to only 9.5% of the 2004 abundance for inside and 41% for outside the no-take zone. Total biomass of clams also decreased inside (from 256 to 32 tonnes) and outside (from 126 to 61 tonnes) the no-take zone. Authors state that decreases were linked with mass-mortalities occurring in 2009 due to a high range of temperature variations. In 2004 a 0.5 km^2 no-take zone was designated for giant clam conservation in Tatakoto Atoll; the rest of the Atoll is open to hand-harvest of clams by local fishers only. Five to six sites inside the no-take zone and 13 sites outside (within the Atoll) were surveyed in 2004 before designation and in 2012. Snorkelers counted and measured all clams inside six 0.25 m^2 quadrats/site (methods fully described in Gilbert *et al*. 2006).

Gilbert A., Andréfouët S., Yan L. & Remoissenet G. (2006). The giant clam *Tridacna maxima* communities of three French Polynesia islands: comparison of their population sizes and structures at early stages of their exploitation. *ICES Journal of Marine Science*, 63, 1573–1589.

A systematic review of 14 studies published until February 2011 of marine protected areas across the world prohibiting all types of fishing (no-take) (24) found that they had more lobsters (species unspecified) compared to marine protected areas only partially prohibiting fishing. Lobster abundance was on average 1.76 times higher in no-take areas compared to partially protected areas (data were not reported, but the analysis outcome was reported as statistical model results). The selected studies compared lobster abundance inside 14 no-take areas with adjacent partial fishing prohibition zones. The abundance data were extracted and used in a meta-analysis.

A paired, site comparison study in 2010–2013 in one soft sediment area in the Firth of Clyde, west coast of Scotland, UK (25) found that five years after designation, a marine reserve closed to all fishing had similar species richness, diversity and overall community composition of large invertebrates compared to an adjacent fished area, but higher species richness, similar diversity, and different overall community composition of small sessile invertebrates. Diversity was reported as diversity indices, and community composition as a graphical analysis. After five years, the closed area had similar total abundance of large invertebrates (60/100 m^2) compared to the fished area (45/100 m^2). However, the closed area had higher abundance of feather stars (closed: 18 vs fished: 11 individuals/100 m^2) and eyelash worms *Myxicola infundibulum* (3 vs 1 individuals/100 m^2). Abundances were similar inside and outside the closed area for large crustaceans and starfish combined (32 vs 29 individuals/100 m^2), and for the parchment worms *Chaetopterus* spp. (6 vs 4 individuals/100 m^2). After five years, species richness of small sessile invertebrates was higher in the closed area (5 species/m^2) than the fished area (4 species/m^2). In addition, abundances (as % cover) were higher inside than outside the protected area for hydroids (inside: 4% vs outside: 3%) and sponges (0.3% vs 0.1%), but abundances were similar inside and outside for bryozoans (0.5%) and for worms, anemones, and tunicates (data not provided). Lamlash Bay marine reserve (2.67 km^2) was established in September 2008 and closed to all fishing. Annually between 2010 and 2013 at 0–30 m depth, 14–20 sites were samples inside and outside the reserve. Large (size unspecified) invertebrates were counted inside 150 m^2 transects. Forty 1 m^2 quadrats/transect were photographed and any small sessile organism (invertebrates including corals and algae) present were identified.

A systematic review of 150 studies published between 1977 and 2012 of marine protected areas prohibiting all fishing (no-take reserves) across the world (26) found that they had more invertebrates compared to fished areas outside. Abundance data were not reported, but the analysis outcome was reported as statistical model results. However, the effect of the reserves on invertebrate abundances differed by trophic group. Abundances were greater inside the reserves compared to outside the reserves for omnivorous and filter-feeding invertebrates, but not for carnivorous and herbivorous invertebrates. The proximity of the reserves to coastal influences (landmasses, urban centres, rivers, populations) did not affect the positive effect of marine reserves on invertebrate abundance. The selected studies compared invertebrate abundances inside and outside 113 no-take reserves. A total of 1,416 abundance comparisons were used in a meta-analysis.

A replicated, randomized, site comparison study in 2000–2014 in two areas of mixed seabed in the northwestern Mediterranean Sea, Spain (27) found that inside a marine protected are prohibiting all fishing (no-take reserve for lobsters) biomass, abundance, and egg production of Caribbean spiny lobsters *Palinurus elephas* were higher, and population size structures were different, compared to a fished area outside the reserve. After prohibiting fishing for 21 to 25 years, lobster biomass and abundance had increased inside the reserve, and were higher compared to outside (data reported as indices). Maximum sizes of lobsters were higher inside the reserve (female: 172; male: 190 mm) compared to outside (female: 130; male: 140). Egg production was higher inside the reserve (3.5 million eggs/unit area) compared to outside (84,000–137,000 eggs/unit area). In 1990, a 55 km^2 marine reserve was established prohibiting all lobster fishing. Lobster catch data between 2000 and 2014 were obtained from fishing nets deployed inside (total number of nets: 252) and at locations outside the reserve (1–150 km away; total number of nets: 2,671).

A replicated, site comparison study in 2013 of twenty sites in the Paterson Inlet, Foveaux Straight, New Zealand (28) found that sites within a marine protected area prohibiting all fishing (no-take reserve) had more New Zealand scallops *Pecten novaezealandiae* after nine years compared to adjacent sites in a recreational harvest-only area, but fewer than at sites in a customary fisheries area. Scallop abundance was higher inside the no-take reserve (0.63 scallops/m^2) compared to the recreational area (0.56 scallops/m^2), but not compare to the customary area (3.62 scallops/m^2). Scallop measured on average 110 mm inside the no-take reserve, 132 mm in the recreational area, and 104 in the customary fisheries area (differences not statistically tested). In June 2013, divers counted and measured scallops in three to nine transects (100 m^2) at each of 20 sites: three in the no-take reserve (designated in 2004), three in the recreational harvest-only area, and six in the customary fisheries area (community-based management, see paper for details).

A site comparison study in 2012–2015 in eight rocky seabed areas in the Firth of Clyde, west coast of Scotland, UK (29) found that the effects of prohibiting all fishing in a marine reserve on the abundances and sizes of European lobsters *Homarus gammarus*, brown crabs *Cancer pagurus* and velvet swimming crabs *Necora puber,* compared to fished areas, varied between the species. After seven years (in 2015) inside the closed area, there were more European lobsters (1 lobster/pot) and velvet swimming crabs (1.3 crab/pot), compared to fished areas (lobsters: 0.4–0.9/pot; crabs: 0.6–0.7/pot), but fewer brown crabs (closed: 0.8/pot; fished: 1.1–1.3/pot). Over the four-year study, the closed area had on average larger lobsters (closed: 100 vs fished: 82–89 mm) and velvet swimming crabs (71 vs 69 mm) compared to the fished area, but smaller brown crabs (125 vs 147 mm). In addition, over the study period, lobster sex ratios (male:female) were similar across all areas, but the closed area had more females with eggs (8/10 females) than fished areas (1/17). Lamlash Bay marine reserve (2.67 km^2) was established in 2008. In 2012–2015, crustaceans were sampled in summer inside the reserve, in three nearby (<2.5 km away) and four distant (10–20 km away; in 2013–2015 only) fished areas outside the reserve, using baited commercial shellfish pots deployed for 48–72 h (three rows of five pots/area) at 0–10 m depth. Organisms caught in pots were identified and counted. Abundance was derived from catch/unit effort. All lobsters, brown crabs, and velvet crabs were measured (carapace length), sexed, and fecundity recorded.

(1) Stoner A.W. & Ray M. (1997) Queen conch, *Strombus gigas*, in fished and unfished locations of the Bahamas: Effects of a marine fishery reserve on adults, juveniles, and larval production. *Fishery Bulletin*, 94, 551–565.

(2) Cole R.G. & Keuskamp D. (1998) Indirect effects of protection from exploitation: Patterns from populations of *Evechinus chloroticus* (Echinoidea) in northeastern New Zealand. *Marine Ecology Progress Series*, 173, 215–226.

(3) Babcock R.C., Kelly S., Shears N.T., Walker J.W. & Willis T.J. (1999) Changes in community structure in temperate marine reserves. *Marine Ecology Progress Series,* 189, 125–134.

(4) Tuya F.C., Soboil M.L. & Kido J. (2000) An assessment of the effectiveness of marine protected areas in the San Juan Islands, Washington, USA. *ICES Journal of Marine Science*, 57, 1218–1226.

(5) Lipcius R.N., Stockhausen W.T. & Eggleston D.B. (2001) Marine reserves for Caribbean spiny lobster: empirical evaluation and theoretical metapopulation recruitment dynamics. *Marine and Freshwater Research,* 52, 1589–1598.

(6) Davidson R.J., Villouta E., Cole R.G. & Barrier R.G.F. (2002) Effects of marine reserve protection on spiny lobster (*Jasus edwardsii*) abundance and size at Tonga Island Marine Reserve, New Zealand. *Aquatic Conservation: Marine and Freshwater Ecosystems,* 12, 213–227.

(7) Rogers-Bennett L., Haaker P.L., Karpov K.A. & Kushner D.J. (2002) Using spatially explicit data to evaluate marine protected areas for abalone in southern California. *Conservation Biology*, 16, 1308–1317.

(8) Shears N.T. & Babcock R.C. (2003) Continuing trophic cascade effects after 25 years of no-take marine reserve protection. *Marine Ecology Progress Series,* 246, 1–16.

(9) Cox C. & Hunt J.H. (2005) Change in size and abundance of Caribbean spiny lobsters *Panulirus argus* in a marine reserve in the Florida Keys National Marine Sanctuary, USA. *Marine Ecology Progress Series*, 294, 227–239.

(10) Parnell P.E., Lennert-Cody C.E., Geelen L., Stanley L.D. & Dayton P.K. (2005) Effectiveness of a small marine reserve in southern California. *Marine Ecology Progress Series*, 296, 39–52.

(11) Pillans S., Pillans R.D., Johnstone R.W., Kraft P.G., Haywood M.D.E. & Possingham H.P. (2005) Effects of marine reserve protection on the mud crab *Scylla serrata* in a sex-biased fishery in subtropical Australia. *Marine Ecology Progress Series,* 295, 201–213.

(12) Shears N.T., Grace R.V., Usmar N.R., Kerr V. & Babcock R.C. (2006) Long-term trends in lobster populations in a partially protected vs. no-take Marine Park. *Biological Conservation*, 132, 222–231.

(13) Pande A., MacDiarmid A.B., Smith P.J., Davidson R.J., Cole R.G., Freeman D., Kelly S. & Gardner J.P. (2008) Marine reserves increase the abundance and size of blue cod and rock lobster. *Marine Ecology Progress Series*, 366, 147–158.

(14) Alexander T.J., Barrett N., Haddon M. & Edgar G. (2009) Relationships between mobile macroinvertebrates and reef structure in a temperate marine reserve. *Marine Ecology Progress Series,* 389, 31–44.

(15) Jack L., Wing S.R. & McLeod R.J. (2009) Prey base shifts in red rock lobster *Jasus edwardsii* in response to habitat conversion in Fiordland marine reserves: implications for effective spatial management. *Marine Ecology Progress Series*, 381, 213–222.

(16) Lester S.E., Halpern B.S., Grorud-Colvert K., Lubchenco J., Ruttenberg B.I., Gaines S.D., Airamé S. & Warner R.R. (2009) Biological effects within no-take marine reserves: a global synthesis. *Marine Ecology Progress Series*, 384, 33–46.
(17a,b) Jack L. & Wing S.R. (2010) Maintenance of old-growth size structure and fecundity of the red rock lobster *Jasus edwardsii* among marine protected areas in Fiordland, New Zealand. *Marine Ecology Progress Series*, 404, 161–172.
(18) Follesa M.C., Cannas R., Cau A., Cuccu D., Gastoni A., Ortu A., Pedoni C., Porcu C. & Cau A. (2011) Spillover effects of a Mediterranean marine protected area on the European spiny lobster *Palinurus elephas* (Fabricius, 1787) resource. *Aquatic Conservation: Marine and Freshwater Ecosystems,* 21, 564–572.
(19) Hoskin M.G., Coleman R.A., Von Carlshausen E. & Davis C.M. (2011) Variable population responses by large decapod crustaceans to the establishment of a temperate marine no-take zone. *Canadian Journal of Fisheries and Aquatic Sciences*, 68, 185–200.
(20) Ling S.D. & Johnson C.R. (2012) Marine reserves reduce risk of climate-driven phase shift by reinstating size-and habitat-specific trophic interactions. *Ecological Applications,* 22, 1232–1245.
(21) Villamor A. & Becerro M.A. (2012) Species, trophic, and functional diversity in marine protected and non-protected areas. *Journal of Sea Research*, 73, 109–116.
(22) Alexander T.J. (2013) Cryptic invertebrates on subtidal rocky reefs vary with microhabitat structure and protection from fishing. *Marine Ecology Progress Series*, 481, 93–104.
(23) Andréfouët S., Van Wynsberge S., Gaertner-Mazouni N., Menkes C., Gilbert A. & Remoissenet G. (2013) Climate variability and massive mortalities challenge giant clam conservation and management efforts in French Polynesia atolls. *Biological Conservation*, 160, 190–199.
(24) Sciberras M., Jenkins S.R., Kaiser M.J., Hawkins S.J. & Pullin A.S. (2013) Evaluating the biological effectiveness of fully and partially protected marine areas. *Environmental Evidence*, 2, 4.
(25) Howarth L.M., Pickup S.E., Evans L.E., Cross T.J., Hawkins J.P., Roberts C.M. & Stewart B.D. (2015) Sessile and mobile components of a benthic ecosystem display mixed trends within a temperate marine reserve. *Marine Environmental Research,* 107, 8–23.
(26) Huijbers C.M., Connolly R.M., Pitt K.A., Schoeman D.S., Schlacher T.A., Burfeind D.D., Steele C., Olds A.D., Maxwell P.S., Babcock R.C. & Rissik D. (2015) Conservation benefits of marine reserves are undiminished near coastal rivers and cities. *Conservation Letters*, 8, 312–319.
(27) Díaz D., Mallol S., Parma A.M. & Goñi R. (2016) A 25-year marine reserve as proxy for the unfished condition of an exploited species. *Biological Conservation*, 203, 97–107.
(28) Twist B.A., Hepburn C.D. & Rayment W.J. (2016) Distribution of the New Zealand scallop (*Pecten novaezealandiae*) within and surrounding a customary fisheries area. *ICES Journal of Marine Science*, 73, 384–393.
(29) Howarth L.M., Dubois P., Gratton P., Judge M., Christie B., Waggitt J.J., Hawkins J.P., Roberts C.M. & Stewart B.D. (2017) Trade-offs in marine protection: Multispecies interactions within a community-led temperate marine reserve. *ICES Journal of Marine Science,* 74, 263–276.

11.3. Designate a Marine Protected Area and prohibit commercial fishing

- **Three studies** examined the effects of prohibiting commercial fishing in marine protected areas on subtidal benthic invertebrates. Two studies were in the South Pacific Ocean[1a,b] (New Zealand), and one in the Caribbean Sea[2] (Mexico).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (3 STUDIES)

- **Crustacean abundance (2 studies):** Two replicated studies (one before-and-after, one site comparison) in the South Pacific Ocean[1a,b] found that after implementing a marine park prohibiting commercial fishing but allowing the recreational harvest of lobsters, lobster abundance inside the park did not increase over the 12 years after implementation[1a], and abundance was similar inside the park and outside where fishing occurred[1b].
- **Crustacean condition (3 studies):** One replicated, before-and-after study in the South Pacific Ocean[1a] found that over the 12 years after implementing a marine park prohibiting commercial fishing but allowing the recreational harvest of lobsters, the biomass of legal-size lobsters inside the park did not increase. One of two site comparison studies (one replicated) in the South Pacific Ocean[1b] and the Caribbean Sea[2] found bigger lobsters in an area closed to commercial fishing

for an unspecified amount of time compared to a fished area[2]. The second study found that 10 years after implementing a marine park prohibiting commercial fishing but allowing the recreational harvest of lobsters, lobster size was similar inside the park and outside where fishing occurred[1b].

BEHAVIOUR (1 STUDY)

- **Crustacean behaviour (1 study):** One site comparison study in the Caribbean Sea[2] found that 80% of the lobster population occurring in a protected area (year of designation unspecified) where commercial fishing was prohibited remained in the unfished area, and thus remained protected.

Background

Commercial fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear entering in contact with the seabed (Collie *et al*. 2000). Specific areas can be designated as protected, and specific management measures taken to control for impactful activities, such as commercial fishing (Villamor & Becerro 2012). Inside protected areas where commercial fishing is prohibited, the threat from commercial fishing to subtidal benthic invertebrates is removed, and previously impacted populations are, in theory, able to recover over time (Ley-Cooper *et al.* 2014). However, species and populations are still subjected to the effects of other fishing activities that are allowed (for instance recreational fishing).

When this intervention occurred outside of a marine protected area, evidence has been summarised under "Threat: Biological resource use – Cease or prohibit commercial fishing". Evidence for interventions related to recreational fishing is summarised under "Threat: Human intrusions and disturbances". Evidence for other interventions related to fisheries restrictions within marine protected areas is summarised under "Habitat protection".

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

Ley-Cooper K., De Lestang S., Phillips B.F. & Lozano-Álvarez E. (2014) An unfished area enhances a spiny lobster, *Panulirus argus*, fishery: implications for management and conservation within a Biosphere Reserve in the Mexican Caribbean. *Fisheries Management and Ecology*, 21, 264–274.

Villamor A. & Becerro M.A. (2012) Species, trophic, and functional diversity in marine protected and non-protected areas. *Journal of Sea Research*, 73, 109–116.

A replicated, before-and-after study in 1977–2005 of nine rocky seabed sites in the South Pacific Ocean, north-eastern New Zealand (1a) found that over the 12 years after implementing a marine park prohibiting commercial fishing but allowing the recreational harvest of spiny lobsters *Jasus edwardsii*, abundance and biomass of lobsters inside the park did not increase. Average lobster abundance was statistically similar before (7–47 lobsters/transect) and 12 years after implementation (4–9). Average biomass of legal-size lobsters (>95 mm carapace length) was similar before (1–3 kg/transect) and after implementation (0–1). Mimiwhangata Marine Park was established in 1984 (implemented 1993). Between 1977 and 2005, nine sites inside the park were surveyed annually. Divers counted all lobsters and visually estimated the size and weight of legal-size lobsters along one 50 x 10 m transect/site.

A replicated, site comparison study in 2003 of 17 rocky seabed sites in the South Pacific Ocean, north-eastern New Zealand (1b) found that 10 years after implementing a marine park prohibiting commercial fishing but allowing the recreational harvest of spiny lobsters *Jasus edwardsii*, abundance and size of lobsters were not higher inside the park compared to outside where fishing occurred. Lobster abundance was not different inside (24 lobsters/transect, of which 8 were legal-sized) and outside the park (28 lobsters/transect, of which 6 were legal-sized). The carapace length of lobsters was not different inside (82 mm) and outside (88 mm) the park. Mimiwhangata Marine Park was established in 1984 (implemented 1993). In 2003, nine sites inside the park and eight fully-fished sites outside were surveyed. Divers counted and visually estimated the size of lobsters along three 50 x 10 m transects/site.

A site comparison study in 2011–2012 of two areas in the Caribbean Sea, Mexico (2) found that Carribbean spiny lobsters *Panulirus argus* grew larger in an area where commercial fishing was banned compared to a fished area, and that a high proportion of the lobster population in the unfished area stayed there over the duration of the study and thus remained protected. Lobster sizes were greater in the unfished area (94 mm) compared to the fished area (73 mm). In the unfished area, this corresponded to 99% of lobsters being bigger than the minimum legal catch size (74.5 mm), while in the fished area it corresponded to only 25%. In addition, an estimated 20% of the lobster population occurring in the unfished area moved to the fished area over the duration of the study, thus 80% remained protected inside the unfished area. The study was carried out in a Biosphere Reserve (year of designation unspecified) which restricted commercial fishing to shallow depths (<20 m) and banned it where depths exceed 20 m (see paper for details). In August–September 2011, lobsters were hand-caught from the unfished area, tagged, sized (carapace length) and released in the unfished area (379 in total). During the 2011/2012 fishing season in the fished area, all lobsters caught by fishermen were sized, and tagged lobsters recorded. A tag-recapture model based on the number of recaptured tagged lobsters (20 in total) was used to estimate the percentage of the lobster population moving from the protected to the fished area.

(1a,b) Shears N.T., Grace R.V., Usmar N.R., Kerr V. & Babcock R.C. (2006) Long-term trends in lobster populations in a partially protected vs. no-take Marine Park. *Biological Conservation*, 132, 222–231.

(2) Ley-Cooper K., De Lestang S., Phillips B.F. & Lozano-Álvarez E. (2014) An unfished area enhances a spiny lobster, *Panulirus argus*, fishery: implications for management and conservation within a Biosphere Reserve in the Mexican Caribbean. *Fisheries Management and Ecology*, 21, 264–274.

11.4. Designate a Marine Protected Area and prohibit bottom trawling

- **Three studies** examined the effects of prohibiting bottom trawling in marine protected areas on subtidal benthic invertebrates. Two studies were in the South Pacific Ocean[1,3] (Australia) and one in the Coral Sea[2] (Australia).

COMMUNITY RESPONSE (3 STUDIES)

- **Overall community composition (2 studies):** One of two replicated, site comparison studies in the South Pacific Ocean[1,3] found that seamounts within a protected area closed to trawling had different invertebrate community composition compared to trawled seamounts and to never-trawled seamounts after four to nine years[3]. The second study found that seamounts within a protected area closed to trawling had different invertebrate community composition compared to shallow unprotected seamounts (heavily trawled) after two years, but not compared to deep unprotected seamounts (lightly trawled)[1].

- **Overall diversity/species richness (3 studies):** One of two replicated, site comparison studies in the South Pacific Ocean[1,3] found that seamounts within a protected area closed to trawling had similar invertebrate species richness and diversity to trawled seamounts and never-trawled seamounts after four to nine years[3]. The second study found that seamounts within a protected area closed to trawling had more invertebrate species compared to shallow unprotected seamounts (heavily trawled) after two years, but not compared to deep unprotected seamounts (lightly trawled)[1]. One randomized, replicated, site comparison study in the Coral Sea[2] found similar combined invertebrate and fish species richness in areas closed to trawling and adjacent fished areas, after seven to eight years.

POPULATION RESPONSE (3 STUDIES)

- **Overall abundance (3 studies):** One of two replicated, site comparison studies in the South Pacific Ocean[1,3] found that seamounts within a protected area closed to trawling had lower invertebrate biomass compared to trawled seamounts and never-trawled seamounts after four to nine years[3]. The second study found that seamounts within a protected area closed to trawling had higher invertebrate biomass compared to shallow unprotected seamounts (heavily trawled) after two years, but not compared to deep unprotected seamounts (lightly trawled)[1]. One randomized, replicated, site comparison study in the Coral Sea[2] found similar invertebrate and fish biomass in areas closed to trawling and adjacent fished areas, after seven to eight years.

Background

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear entering in contact with the seabed (Collie *et al*. 2000). Mobile fishing gear such as bottom trawls are known to be particularly damaging as they are dragged along/above the seabed, but can be prohibited within an area. Specific areas can be designated as protected, and specific management measures taken to control for impactful activities, such as bottom trawling (Huvenne *et al.* 2016). Inside protected areas where bottom trawling is prohibited, the threat from bottom trawling to subtidal benthic invertebrates is removed, and previously impacted populations are, in theory, able to recover over time (Hiddink *et al.* 2017). However, species and populations are still subjected to the effects of other fishing activities allowed (for instance commercial potting or recreational fishing).

When this intervention occurred outside of a marine protected area, evidence has been summarised under "Threat: Biological resource use – Cease or prohibit bottom trawling". Evidence for other related interventions is summarised under "Threat: Biological resource use".

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

Hiddink J.G., Jennings S., Sciberras M., Szostek C.L., Hughes K.M., Ellis N., Rijnsdorp A.D., McConnaughey R.A., Mazor T., Hilborn R. & Collie J.S. (2017) Global analysis of depletion and recovery of seabed biota after bottom trawling disturbance. *Proceedings of the National Academy of Sciences,* 114, 8301–8306.

Huvenne V.A.I., Bett B.J., Masson D.G., Le Bas P. & Wheeler A.J. (2016) Effectiveness of a deep-sea cold-water coral Marine Protected Area, following eight years of fisheries closure. *Biological Conservation*, 200, 60–69.

A replicated, site comparison study in 1997 of 14 seamounts south of Tasmania, South Pacific Ocean, Australia (1) found that seamounts within a protected area closed to trawling tended to have different invertebrate community composition, more species and higher biomass of invertebrates, compared to shallow unprotected seamounts, but not compared to deep unprotected seamounts, after two years. Results were not tested for statistical significance. Invertebrate community composition appeared typically similar at protected seamounts and deep unprotected seamounts, but different to that of shallow unprotected seamounts (data presented as graphical analyses). Protected seamounts tended to have more invertebrate species (22 species/sample) and biomass (6 kg/sample) compared to shallow unprotected seamounts (9 species/sample; 1 kg/sample) and similar to deep unprotected seamounts (20 species/samples; 7 kg/sample). The low diversity and biomass at shallow unprotected were associated with the loss of coral substrate from intense trawling. In 1995, a protected area was established and closed to trawling. In 1997, invertebrates (including corals) (>25 mm) living on the seamounts inside (6 seamounts; 12 samples) and outside (8 seamounts; 22 samples) the protected area (peaks at approximately 660–1,700 m depths) were sampled using a dredge. Invertebrates were sorted into groups and weighed by groups. Shallow unprotected seamounts were heavily fished, but deep seamounts were only lightly fished.

A randomized, replicated, site comparison study in 1992–1993 in four areas of mixed seabed inside the Great Barrier Reef Marine Park off northern Queensland, Coral Sea, Australia (2) found no difference in the biomass of non-commercial unwanted catch (invertebrates and fish discard) or in the number of 'common' and 'rare' discard species between areas closed to trawling and adjacent fished areas, seven to eight years after the closure. Data were reported as statistical model results. A 10,000 km^2 area of the Great Barrier Reef Marine Park was closed to trawling in 1985. Two surveys were carried out, one in 1992 and one in 1993. During each survey, 25 randomly selected sites were sampled at each of four areas within the marine park, two closed areas, and two fished areas located 10 nm away, using both a benthic dredge and a prawn trawl. A total of 156 dredges (86 in closed areas, 70 in fished areas) and 122 trawls (68 in closed areas, 54 in fished areas) were towed. For each tow, discard species were collected, identified, counted, and weighed from subsamples (amount not specified). Total weight of discard was estimated from the subsamples. Species were either recorded as 'common' (found in at least 11 of the 25 sites) or 'rare' (found in 10 or fewer sites).

A replicated, site comparison study in 2006 of 25 deep-sea seamounts located south of Tasmania, South Pacific Ocean, Australia (3) found that, four to nine years after prohibiting bottom trawling in marine protected areas, invertebrate community composition was different and abundance lower at protected seamounts compared to trawled and natural (never trawled) seamounts, and diversity and species richness was similar to trawled but lower than at natural seamounts. Community data were reported as graphical analyses and diversity data as diversity indices. Species richness was similar at protected (46 species/1,270 m^2) and trawled seamounts (46), but lower than natural seamounts (52). Abundance was lowest at protected (1–3 individuals/m^2), compared to trawled seamounts (3–5), and natural seamounts where abundance was the highest (5–18). Species richness, diversity, and abundance were positively related to the cover of habitat-forming corals, which was higher on protected seamounts (3%) than trawled seamounts (0.1%), but lower than on natural seamounts (52%). Invertebrates (including corals) were identified and counted at 25 seamounts from videos transects (up to 4.7 km long, from 1,100 to 1,400 m depth; 38 transects in total). Ten seamounts were located either in continuously trawled areas or in areas where trawling had stopped following establishment of reserves (at some point between 1997 and 2003), and 15 were in never-trawled natural areas. Fishing history of individual seamounts was verified using logbook data from the Australian Fisheries Management Authority.

(1) Koslow J.A., Gowlett-Holmes K., Lowry J.K., O'Hara T., Poore G.C.B. & Williams A. (2001) Seamount benthic macrofauna off southern Tasmania: Community structure and impacts of trawling. *Marine Ecology Progress Series,* 213, 111–125.
(2) Burridge C.Y., Pitcher C.R., Hill B.J., Wassenberg T.J. & Poiner I.R. (2006) A comparison of demersal communities in an area closed to trawling with those in adjacent areas open to trawling: a study in the Great Barrier Reef Marine Park, Australia. *Fisheries Research*, 79, 64–74.
(3) Althaus F., Williams A., Schlacher T.A., Kloser R.J., Green M.A., Barker B.A., Bax N.J., Brodie P. & Schlacher-Hoenlinger M.A. (2009) Impacts of bottom trawling on deep-coral ecosystems of seamounts are long-lasting. *Marine Ecology Progress Series,* 397, 279–294.

11.5. Designate a Marine Protected Area and install physical barriers to prevent trawling

- **One study** examined the effects of installing physical barriers to prevent trawling in a protected area on subtidal benthic invertebrate populations. The study was in the South China Sea[1] (Hong Kong).

COMMUNITY RESPONSE (1 STUDY)

- **Worm community composition (1 study):** One replicated, site comparison study in the South China Sea[1] found that sites in a protected area where physical barriers were installed to prevent trawling had a different community composition of nematode worms compared to nearby unprotected fished sites, after up to two years.
- **Worm species richness/diversity (1 study):** One replicated, site comparison study in the South China Sea[1] found that sites in a protected area where physical barriers were installed to prevent trawling had similar diversity and species richness of nematode worms to nearby unprotected fished sites, after up to two years.

POPULATION RESPONSE (1 STUDY)

- **Overall abundance (1 study):** One replicated, site comparison study in the South China Sea[1] found that sites in a protected area where physical barriers were installed to prevent trawling had fewer small invertebrates compared to nearby unprotected fished sites, after up to two years.
- **Worm abundance (1 study):** One replicated, site comparison study in the South China Sea[1] found that sites in a protected area where physical barriers were installed to prevent trawling had fewer nematode worms compared to nearby unprotected fished sites, after up to two years.

Background

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear entering in contact with the seabed (Collie *et al*. 2000). Some habitats, such as coral reefs and seagrass meadows, are particularly vulnerable to trawling gears. Specific areas can be designated as protected, and specific management measures taken to control for impactful activities (Kelleher 1999). In marine protected areas where trawling is prohibited, physical barriers, such as concrete blocks or other artificial reefs, can be placed to ensure no illegal trawling takes place, as such physical barriers would damage trawl nets (Liu *et al*. 2011).

When this intervention occurs outside of a protected area, evidence has been summarised under "Threat: Biological resource use – Install physical barriers to prevent illegal trawling". Evidence for related interventions is summarised under "Habitat restoration and creation – Create artificial reefs".

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

Kelleher G. (1999) Guidelines for marine protected areas. IUCN, Gland, Switzerland and Cambridge, UK.
Liu, X.S., Xu W.Z., Cheung S.G. & Shin P.K.S. (2011) Response of meiofaunal community with special reference to nematodes upon deployment of artificial reefs and cessation of bottom trawling in subtropical waters, Hong Kong. *Marine Pollution Bulletin,* 63, 376–384.

A replicated, site comparison study in 2007–2008 of four soft seabed sites in the South China Sea, Hong Kong (1) found that sites inside a marine protected area where barriers were deployed to prevent trawling had fewer small invertebrates and nematode worms, a different nematode community composition, but similar nematode diversity and species richness, compared to adjacent unprotected fished sites, after up to two years. Invertebrate abundance was lower in the protected area (198 individuals/10 cm^2), compared to the unprotected fished area (290 individuals/10 cm^2). Nematode abundance was lower in the protected area (183 individuals/10 cm^2), compared to the unprotected fished area (280 individuals/10 cm^2). Nematode community was different inside and outside the protected area (community data reported as a graphical analysis). Nematode diversity (reported as diversity indices) and species richness were typically similar in the protected (ranging from 10 to 39 species) and unprotected areas (ranging from 20 to 33 species). Increased abundances were associated with increased sediment disturbance from trawling. In 2006, a 12 km^2 area was designated as a marine protected area and trawling discouraged by placing artificial reefs and concrete blocks with steel spikes inside and along the boundary of the area. Two sites inside and two outside the protected area were sampled at quarterly intervals between April 2007 and January 2008 at 20 m depth using a sediment grab (0.1 m^2). Small invertebrates (0.038–0.5 mm) were extracted and counted. Nematode worms were identified and counted. In the sites outside the protected area, one bottom trawl was conducted prior to each sampling.

(1) Liu, X.S., Xu W.Z., Cheung S.G. & Shin P.K.S. (2011) Response of meiofaunal community with special reference to nematodes upon deployment of artificial reefs and cessation of bottom trawling in subtropical waters, Hong Kong. *Marine Pollution Bulletin,* 63, 376–384.

11.6. Designate a Marine Protected Area and prohibit dredging

- **One study** examined the effects of prohibiting dredging in marine protected areas on subtidal benthic invertebrates. The study was in the Firth of Lorn[1] (UK).

COMMUNITY RESPONSE (1 STUDY)

- **Overall community composition (1 study):** One paired, replicated, site comparison study in the Firth of Lorn[1] found that sites inside a protected area that had been prohibiting dredging for approximately 2.5 years had different combined invertebrate and fish community composition compared to unprotected dredged sites.

POPULATION RESPONSE (1 STUDY)

- **Overall abundance (1 study):** One paired, replicated, site comparison study in the Firth of Lorn[1] found that sites inside a protected area that had been prohibiting dredging for approximately 2.5 years typically had greater combined cover of bryozoans and hydroids (combined) compared to unprotected dredged sites.

Background

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear entering in contact with the seabed (Collie *et al.* 2000). Mobile fishing gear such as towed dredges, involve towing a heavy steel frame along the bottom of the seabed, and as such are known to be particularly damaging. They are used for instance in the harvest of bivalves (e.g. mussels, clams, scallops). Recreational and artisanal bivalve fishing may cause less impact due to the smaller scale of the operations and different harvesting methods used (for instance hand-harvest). Specific areas can be designated as protected, and specific management measures taken to control scallop dredging (Blyth *et al*. 2002; Boulcott *et al*. 2014). Inside protected areas where dredging is prohibited, the threat from dredging to subtidal benthic invertebrates is removed, and previously impacted populations are, in theory, able to recover over time (Blyth *et al.* 2004). However, species and populations are still subjected to the effects of other fishing activities allowed (for instance commercial potting or recreational fishing).

Evidence for related intervention is summarised under "Habitat protection – Designate a Marine Protected Area and prohibit the harvest of scallops", "Designate a Marine Protected Area and prohibit all towed (mobile) fishing gear" and "Species management – Cease or prohibit the harvest of scallops". When this intervention occurred outside of a marine protected area, evidence has been summarised under "Threat: Biological resource use – Cease or prohibit dredging".

Blyth R.E., Kaiser M.J., Edwards-Jones G. & Hart P.J.B. (2002) Voluntary management in an inshore fishery has conservation benefits. *Environmental Conservation*, 29, 493–508.

Boulcott P., Millar C.P. & Fryer R.J. (2014) Impact of scallop dredging on benthic epifauna in a mixed-substrate habitat. *ICES Journal of Marine Science,* 71, 834–844.

Bull M.F. (1989) *The New Zealand scallop fishery: a brief review of the fishery and its management*. Edited by: MLC Dredge, WF Zacharin and LM Joli, 42.

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

Schejter L., Bremec C.S. & Hernández D. (2008) Comparison between disturbed and undisturbed areas of the Patagonian scallop (*Zygochlamys patagonica*) fishing ground "Reclutas" in the Argentine Sea. *Journal of Sea Research*, 60, 193–200.

A paired, replicated, site comparison study in 2009 of nine sites in three areas of sandy and rocky seabed in the Firth of Lorn, Scotland, UK (1) found that sites inside a protected area that had been prohibiting dredging for approximately 2.5 years typically had greater combined cover of bryozoans and hydroids and different combined invertebrate and fish community composition compared to unprotected dredged sites. In four of six comparisons, sites inside the protected area had higher cover of bryozoans and hydroids compared to dredged sites outside (inside 43 vs outside 34%; 25 vs 15%; 21 vs 10%; 22 vs 9%). In two comparisons, cover was similar inside and outside (19 vs 14%; 52 vs 54%). Community composition varied across the three areas, but within each area was always different in the protected and dredged sites (data presented as graphical analyses). Part of the Firth of Lorn was designated as a Special Area of Conservation in March 2005 and closed to scallop dredging in the boreal spring of 2007. Three areas (25–89 m depth) along the boundary of the closed area were selected. In each area, one site on each side of the boundary (i.e. one inside the close area, one outside) was surveyed in May and again in July–August 2009. Invertebrates were surveyed using a camera at 30–40 sampling stations/area. For three photographs/station/survey, the combined area covered by erect bryozoans and hydroids was measured, and all animals (24 invertebrate species; combined with one species of skate and one group of fish species) identified and counted.

(1) Boulcott P., Millar C.P. & Fryer R.J. (2014) Impact of scallop dredging on benthic epifauna in a mixed-substrate habitat. *ICES Journal of Marine Science,* 71, 834–844.

11.7. Designate a Marine Protected Area and prohibit all towed (mobile) fishing gear

- **Two studies** examined the effects of prohibiting all towed gear in marine protected areas on subtidal benthic invertebrate populations. One study was in the Bristol Channel and the Irish Sea[1] (UK), the other in the English Channel[2] (UK).

COMMUNITY RESPONSE (1 STUDY)

- **Overall community composition (1 study):** One before-and-after, site comparison study in the English Channel[2] found that, over the three years after closing a marine protected area to all towed gears, the community composition of reef-indicative invertebrate species became different to that of unprotected fished sites.
- **Overall diversity/species richness (1 study):** One before-and-after, site comparison study in the English Channel[2] found that, over the three years after closing a marine protected area to all towed gears, the number of reef-indicative invertebrate species remained similar to unprotected fished sites.

POPULATION RESPONSE (2 STUDIES)

- **Overall abundance (1 study):** One before-and-after, site comparison study in the English Channel[2] found that, over the three years after closing a marine protected area to all towed gears, the abundance of reef-indicative invertebrate species became greater than at unprotected fished sites.
- **Crustacean abundance (1 study):** One replicated, site comparison study in the Bristol Channel and the Irish Sea[1] found that a marine protected area closed to all towed gear (only allowing potting) for 33 to 36 years had mixed effects on the abundances of lobsters and crabs depending on species.
- **Crustacean condition (1 study):** One replicated, site comparison study in the Bristol Channel and the Irish Sea[1] found that a marine protected area closed to all towed gear (only allowing potting) for 33 to 36 years had mixed effects on the sizes of lobsters and crabs depending on species.

Background

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear entering in contact with the seabed (Collie *et al*. 2000). Mobile fishing gear such as bottom trawls, dredges, and other towed gears, are known to be particularly damaging as they are dragged onto the seabed. Specific areas can be designated as protected, and specific management measures taken to control for towed gears (Blyth *et al*. 2002; Sheehan *et al.* 2013). Inside protected areas where mobile fishing gear is prohibited, the threat from these practices to subtidal benthic invertebrates is removed, and previously impacted populations are, in theory, able to recover over time (Blyth *et al*. 2004). However, species and populations are still subjected to the effects of other fishing activities allowed (for instance commercial potting or recreational fishing).

When this intervention occurred outside of a protected area, evidence has been summarised under "Threat: Biological resource use - Cease or prohibit all towed (mobile) fishing gear". Evidence for related interventions is summarised under "Habitat protection – Designate a Marine Protected Area and prohibit bottom trawling" and "Designate a Marine Protected Area and prohibit dredging".

Blyth R.E., Kaiser M.J., Edwards-Jones G. & Hart P.J.B. (2004) Implications of a zoned fishery management system for marine benthic communities. *Journal of Applied Ecology*, 41, 951–961.
Blyth R.E., Kaiser M.J., Edwards-Jones G. & Hart P.J.B. (2002) Voluntary management in an inshore fishery has conservation benefits. *Environmental Conservation*, 29, 493–508.
Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.
Sheehan E.V., Cousens S.L., Nancollas S.J., Stauss C., Royle J. & Attrill M.J. (2013) Drawing lines at the sand: Evidence for functional vs. visual reef boundaries in temperate Marine Protected Areas. *Marine Pollution Bulletin*, 76, 194–202.

A replicated, site comparison study in summer 2004–2007 of eight sites in four areas of rocky and sandy seabed in the Bristol Channel and the Irish Sea, UK (1) found that a marine protected area closed to all towed gear for 33–36 years had mixed effects on the abundances and sizes of European lobster *Homarus Gammarus,* and three crab species. Abundances of large lobsters (≥90 mm) did not change over time in any areas, where they were similar (1–2 lobsters/line). Abundance of small lobsters (<90 mm) increased in the protected areas by 140% (due to spill-over effects from an adjacent no-take zone; from 2 to 4–7 lobsters/line), but not in the fished areas where abundance remained constant (2–4 lobsters/line). The size of large lobsters (≥90 mm) decreased similarly in all areas by 2–3% (from 98 to 95 mm). Abundance of velvet crabs *Necora puber* decreased inside the protected areas (from 5–6 to 1 crab/line) but increased in the fished areas (from 0–6 to 1–7 crabs/line). Abundance of brown crabs *Cancer pagarus* did not change over time in any areas but was on average higher in the protected areas (1–2 crabs/line) compared to the fished areas (0.3–2 crabs/line). The average size of brown crabs did not change over time in any areas, and was not different between protected (123–128 mm) and fished areas (116–130 mm). Abundance of spider crabs *Maja squinado* was similar in 2004 and 2007 for all areas but varied spatially. Lundy Island marine protected area was designated as a voluntary reserve in 1971 (statutory since 1986) and only allowed crab and lobster potting (all other fishing prohibited; apart from a small 4 km^2 no-take zone). In 2004–2007, lobsters and crabs were surveyed at two locations in the protected area (outside the no-take zone), and two unprotected fished locations (20–100 km away) (2 sites/location). Four lines of standard commercial baited shellfish pots were deployed (10 pots/line) at each site for 24 h. Upon retrieval, lobsters and crabs were counted and measured (carapace length). The pots were redeployed for five consecutive days each year.

A before-and-after, site comparison study from 2008–2011 of 12 sites of rocky reefs and pebbly sand seabed in Lyme Bay, English Channel, southwest England, UK (2) found that three years after closing a marine protected area to all towed gears, community composition and abundance of reef-indicative invertebrate species became different to unprotected fished sites, but their species richness remained similar. Community data were reported as graphical analyses. Reef-indicative invertebrate species richness did not statistically change over time and was similar in closed and fished sites both before closure (closed: 8; fished: 5 species/m^2) and three years after (closed: 10; fished: 5 species/m^2). However, while before closure their abundance was similar in closed (6 individuals/m^2) and fished sites (3 individuals/m^2), it increased over time in closed sites and was greater than in fished sites after three years (closed: 16; fished: 1 individuals/m^2). In particular, abundances of four key species increased in closed sites over time and became more abundant than in fished sites after three years (significantly for: bryozoans *Pentapora fascialis* closed 0.11 vs fished 0.01; sea squirts *Phallusia mammillata* 0.06 vs 0.01 and branching sponges 0.05 vs <0.01; non-significantly for hydroids 55 vs 17 individuals/m^2). The 206 km^2 protected area was closed to towed fishing gears in 2008. Six weeks after closure (considered the 'before' timepoint by the authors) and in 2011, five sites inside and seven sites outside the protected area were surveyed using a video camera (two 200 m video-transects/site). All invertebrates observed on the video present on pebbly sand, but indicative of rocky reef habitat, were identified and counted.

(1) Hoskin M.G., Coleman R.A., Von Carlshausen E. & Davis C.M. (2011) Variable population responses by large decapod crustaceans to the establishment of a temperate marine no-take zone. *Canadian Journal of Fisheries and Aquatic Sciences*, 68, 185–200.
(2) Sheehan E.V., Cousens S.L., Nancollas S.J., Stauss C., Royle J. & Attrill M.J. (2013) Drawing lines at the sand: Evidence for functional vs. visual reef boundaries in temperate Marine Protected Areas. *Marine Pollution Bulletin*, 76, 194–202.

11.8. Designate a Marine Protected Area with a zonation system of activity restrictions

- **Thirteen studies** examined the effects of designating a marine protected area with a zonation system of activity restrictions on subtidal benthic invertebrate populations. Four studies were in the Caribbean Sea[1,2,11,13] (Belize, Mexico), three in the Mediterranean Sea[3–5] (Italy), one in the Central Pacific Ocean[6] (Ecuador), three in the Bristol Channel and the Irish Sea[7,9,10] (UK), one in the Indian Ocean[8] (Australia), and one in the North Atlantic Ocean[12] (Portugal).

COMMUNITY RESPONSE (2 STUDIES)

- **Overall community composition (1 study):** One site comparison study in the Mediterranean Sea[3] found that inside a marine protected area with a zonation system, the combined invertebrate and algae species community composition was different at a site prohibiting all fishing compared to sites where some fishing occurs, after six years.
- **Overall species richness/diversity (1 study):** One site comparison study in the North Atlantic Ocean[12] found that inside a marine protected area with a zonation system, sites prohibiting nearly all fishing had similar invertebrate species richness to sites where fishing was mostly allowed, after two years.

POPULATION RESPONSE (13 STUDIES)

- **Overall abundance (1 study):** One site comparison study in the North Atlantic Ocean[12] found that inside a marine protected area with a zonation system, abundances of specific invertebrate groups varied between sites prohibiting nearly all fishing and sites where fishing was mostly allowed, after two years.
- **Crustacean abundance (7 studies):** Three of seven site comparison studies (two replicated) in the Caribbean Sea[1,2,13], the Central Pacific Ocean[6], and in the Bristol Channel and the Irish Sea[7,9,10] found that inside a marine protected area with a zonation system, abundance and/or biomass of spiny lobsters increased in a zone closed to all/commercial fishing and were greater than in a zone where fewer fishing restrictions occurred[1,2,13], after four to 20 years depending on the study. One found that sites closed to all fishing had higher abundances of spiny lobsters and slipper lobsters after eight to ten years compared to fished sites[6]. Two found that sites closed to all fishing for six to seven years had more European lobsters than sites where potting was allowed[9,10]. And one found that abundances of European lobsters, velvet crabs, brown crabs and spider crabs, after one to four years, varied with the levels of protection[7].
- **Crustacean condition (4 studies):** Three of five site comparison studies (one replicated) in the Bristol Channel and the Irish Sea[7,9,10], and in the Caribbean Sea[11,13] found that, inside a marine protected area with a zonation system, sites prohibiting all fishing for seven years[9,10] or

commercial fishing (duration unspecified)[11] had bigger lobsters compared to fished areas. One found that the sizes of lobsters, velvet crabs, brown crabs and spider crabs varied with the levels of protection[7], and one study found that the size of spiny lobsters decreased similarly in an area prohibiting all fishing and in an area with fewer restrictions 14 to 20 years after designation of the protected area[13]. Two studies undertaken in the same area found conflicting effects of prohibiting all fishing for six to seven years on disease and injury of lobsters[9,10].

- **Echinoderm abundance (2 studies):** One of two site comparison studies in the Mediterranean Sea[4,5] found that inside a marine protected area with a zonation system, at a site prohibiting all fishing for 17 to 18 years, abundances of two species of sea urchins were higher than at sites allowing the recreational fishing of purple sea urchins[4].The other one found similar abundance of purple sea urchins inside fully protected sites, sites where some restricted urchin harvest occurs, and unprotected fished sites outside the protected area after five years[5].
- **Echinoderm condition (2 studies):** Two site comparison studies in the Mediterranean Sea[4,5] found that inside a marine protected area with a zonation system, sites prohibiting all fishing had bigger sea urchins compared to sites where some restricted urchin harvest occurs[4,5] and compared to unprotected fished sites outside the protected area[5], after either four years[5] or 17 to 18 years[4].
- **Mollusc abundance (3 studies):** One replicated, randomized, controlled study in the Indian Ocean[8] found that inside a marine protected area with a zonation system, abundance of blacklip abalone was higher in sites that had been prohibiting all fishing for five years compared to those prohibiting commercial fishing only. Two site comparison studies in the Caribbean Sea[1,13] found that inside marine protected areas with a zonation system, abundances of adult queen conch increased over time in a zone closed to all fishing and were greater than in zones with fewer restrictions, but abundances of juvenile conch did not differ or vary differently between zones, after either five to eight years[1] or 14 to 20 years[13].
- **Mollusc condition (1 study):** One site comparison study in the Caribbean Sea[13] found that inside a marine protected area with a zonation system, the size of queen conch decreased similarly in the area prohibiting all fishing and in the area with fewer restrictions, after 14 to 20 years.
- **Sponge abundance (1 study):** One site comparison study in the Mediterranean Sea[3] found that inside a marine protected area with a zonation system, the cover of sponges *Cliona* spp. was higher at a site prohibiting all fishing for six years compared to sites where some fishing occurred.

BEHAVIOUR (1 STUDY)

- **Crustacean behaviour (1 study):** One site comparison study in the Caribbean Sea[11] found that, inside a marine protected area with a zonation system (year of designation unspecified), 80% of the lobster population occurring in the unfished area remained in the protected unfished area, and thus remained protected.

Background

Fishing and other anthropogenic activities can impact subtidal benthic invertebrates through species removal or habitat damage, for example from gear entering in contact with the seabed (Collie *et al*. 2000). Specific areas can be designated as protected, and specific management measures taken to control for impactful activities, such as commercial, recreational, or artisanal fishing (Villamor & Becerro 2012), or restricting specific gear or practices. Some protected areas are designed with a zonation system, whereby specific areas are designed as “no-take zones” prohibiting all fishing and activities, others within the same protected area are designated as “refuge” or “restricted” zones and restrictions vary (some activities are allowed), and even “general use” or “buffer” zones whereby nearly all activities are allowed. This type of marine protected area design is fairly common in certain areas, such as in the Mediterranean Sea (Francour *et al*. 2001).

Inside protected areas with a zonation system, levels of restriction can benefit previously impacted populations which are, in theory, able to recover over time and can even spill-over from the most protected zones into the less protected zones (Hoskin *et al.* 2011). However, species and populations are still subjected to the effects of other fishing activities allowed (for instance recreational fishing).

Evidence for related interventions on fishing restrictions within marine protected areas is summarised under “Habitat protection”.

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

Francour P., Harmelin J.G., Pollard D. & Sartoretto S. (2001) A review of marine protected areas in the northwestern Mediterranean region: siting, usage, zonation and management. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 11, 155–188.

Hoskin M.G., Coleman R.A., Von Carlshausen E. & Davis C.M. (2011) Variable population responses by large decapod crustaceans to the establishment of a temperate marine no-take zone. *Canadian Journal of Fisheries and Aquatic Sciences*, 68, 185–200.

Villamor A. & Becerro M.A. (2012) Species, trophic, and functional diversity in marine protected and non-protected areas. *Journal of sea research*, 73, 109–116.

A site comparison study from 1997–2001 of two coral reef, seagrass and sandy seabed areas in Glover’s Reef marine reserve, western Caribbean Sea, off the coast of Belize (1) found that between five and eight years after designating a marine protected area with a zonation system, abundances of adult spiny lobster *Panulirus argus* and adult queen conch *Strombus gigas* increased in a zone closed to all fishing and were greater than in a zone where commercial fishing occurred. Abundance of adult lobsters (>45 mm carapace length) increased in the closed zone (after four years in 1997: 21; after eight years in 2001: 84 lobsters/ha) and was greater than in the fished zone where abundance did not change (1997: 13; 2001: 26). Abundance of adult conch (>110 mm shell length) increased in the closed zone (1997: 244, 2001: 921 conch/ha), and was greater than in the fished zone where abundance did not change (1997: 296, 2001: 188). Abundance of juvenile lobsters and conch did not vary over time or differ between zones. The reserve was established in 1993, with a general use zone open to commercial fishing and a zone prohibiting all fishing. In each zone inside the reserve, divers counted and measured lobsters (in eight 0.5 ha coral reef patches/zone) and conch (in twenty-four 200 m^2 transects on sand/zone) at quarterly intervals in 1997–2001.

A site comparison study from 1996–2001 of two coral reef, seagrass, and sandy seabed areas at Glover’s Reef marine reserve, western Caribbean Sea, off the coast of Belize (2) found that between four and eight years after designating a marine protected area with a zonation system, abundance and biomass of Caribbean spiny lobster *Panulirus argus* increased in a zone closed to all commercial fishing and were also greater than in a zone where artisanal commercial fishing occurred. Annual average abundance of lobsters increased in the closed zone (after three years in 1996: 63, after eight years in 2001: 144 lobsters/ha), and was greater than in the fished zone where abundance did not change (1996: 42; 2001: 61 lobsters/ha). Biomass of adult lobsters (>76 mm carapace length) increased in the closed zone (1996: 10; 2001: 155 kg/ha) and was higher than in the fished zone where biomass did not change (1996: 7; 2001: 3.5 kg/ha). All data were extracted from the graphs presented in the study. The reserve was established in 1993, with a general use zone open to artisanal commercial fishing and a zone prohibiting all commercial fishing. In each zone inside the reserve, divers counted and measured lobsters in eight 0.007–0.5 ha coral reef patches/zone at quarterly intervals in 1996–2001.

A site comparison study in 2002–2003 of three rocky sites within a marine protected area with a zonation system in northeast Sardinia, Mediterranean Sea, Italy (3) found that a fully protected site prohibiting all fishing had a different invertebrate and algae (species combined) community composition to partially protected sites where some fishing occurred, after six years of enforcement. Community data were reported as graphical analyses. In addition, the cover of sponges *Cliona* spp. was higher in the fully protected site (6–12 % cover) compared to the partially protected sites (1–5 % cover). In September 2002 and 2003, algae and invertebrates were surveyed once at three sites (5 m depth) inside a protected area: a no-take zone (one site) and a partially protected zone where some regulated fishing takes places (two sites). Restrictions and limitations had been enforced since 1997. Divers photographed 10 quadrats (16 × 23 cm)/site. Percentage cover of sessile invertebrates and algae were estimated from photographs.

A site comparison study in 2003–2004 of three rocky seabed sites within a marine protected area with a zonation system, off Ustica Island, Mediterranean Sea, Italy (4) found that overall at a fully protected site that had been prohibiting all fishing for 17–18 years, abundances and sizes of two species of sea urchins were higher than at partially protected sites where recreational fishing of purple sea urchins *Paracentrotus lividus* occurred, but effects varied seasonally. Abundances of purple and black sea urchins *Arbacia lixula* were higher in the fully protected than the partially protected sites in summer (purple: 2.9 vs 0.7–1.3/m^2, black: 3.1 vs 1.7–1.9/m^2) and autumn (purple: 4.1 vs 1.6–2.1/m^2, black: 2.3 vs 0.7–1/m^2), but not spring (purple: 2.3 vs 2.3–2.9/m^2, black: 1.5 vs 2.0/m^2). Purple sea urchins were larger in the fully protected than the partially protected sites in spring (fully: 45 vs partially: 31–34 mm), summer (43 vs 35–37 mm) and autumn (44 vs 32–35 mm). Black sea urchins were smaller in the fully protected than the partially protected sites in autumn (31 vs 35–37 mm), but similarly sized across sites in spring and summer (37–42 vs 39–45 mm). Ustica Island marine protected area was established in 1986 with a no-take zone and a partially protected zone where some recreational activities take place. In 1994, recreational fishing for purple sea urchin inside the partially protected zone was reopened following undesirable increases in their abundance leading to barren areas. At one site in the no-take zone and two in the partially protected (4–8 m depth), divers identified and counted all urchins along three 10 m^2 transects, twice in autumn 2003, in spring 2004, and in summer 2004. The diameter (not including spines) of urchins inside 1 m^2 quadrats was measured.

A site comparison study in 2006 of seven sites in a seagrass and rocky seabed area in the Mediterranean Sea, Sardinia, Italy (5) found that the effect of designating a marine protected area with a zonation system on purple sea urchin *Paracentrotus lividus* abundance and size varied with the level of restriction in place. Within the protected area after four years, fully protected no-take sites had similar abundances of urchin (2–5 individuals) compared to partially protected sites where some restricted urchin harvest occurred (1–12 individuals), and to unprotected fished sites outside the protected area (2–12 individuals). However, urchins were larger in no-take sites (57–62 mm), compared to partially protected (32–61 mm) and unprotected fished sites (24–50 mm). Capo Caccia–Isola Piana marine protected area was established in 2002 with varying levels of protection including a no-take zone and a partially protected zone where urchin harvest was formerly prohibited but reopened with restrictions in 2006 (see paper for details). Sampling took place in April–May 2006 after the harvest season at seven sites (200 m^2 each) in 6–10 m water depth: one within the no-take zone, three within the partially protected zone, and three outside the marine protected area. At each site, urchins were counted inside 20 quadrats (1 × 1 m), and 20 urchins were measured (diameter without spines).

A replicated, site comparison study in 2000–2002 of 20 rocky seabed sites inside the Galapagos Marine Reserve, eastern Pacific Ocean, Ecuador (6) found that protected sites that had been closed to all fishing for eight to ten years had higher abundances of spiny lobsters *Panulirus penicillatus* and slipper lobsters *Scyllarides astori,* compared to fished sites inside the reserve. Encounter rates (indicative of abundance) of spiny lobster and slipper lobster were higher in the closed areas (spiny: 0.4; slipper: 0.2 lobsters/hr) than the fished areas (spiny: 0.1; slipper: 0.1 lobsters/hr). Pencil urchin *Eucidaris galapagensis* abundance was lower in closed areas (2.2 urchins/m^2) than fished areas (4.5 urchins/m^2). Fishing exclusion zones within the reserve were created in 1992 and formally established in 2000, but uneven compliance with the fishing regulations is reported. In April 2000–August 2002, divers surveyed lobsters and sea urchins at ten sites within exclusion zones and ten sites outside (but inside the reserve). Lobsters were counted along four 20-min dive transects/site.

A replicated, site comparison study in summer 2004–2007 of ten rocky and sandy sites, across two zones inside a marine protected area and two areas outside, in the Bristol Channel and the Irish Sea, UK (7) found that abundances and sizes of European lobster *Homarus gammarus* and three crab species varied with the levels of protection. Abundance of large lobsters (≥90 mm) increased by 127% inside the no-take zone between 2004 and 2007 (one to four years after designation of the no-take zone; from 3 to 7 lobsters/line) and was five times higher than in aa partly fished zone (potting only) inside the protected area and fully fished areas outside where abundance had not changed (1–2 lobsters/line). Abundance of small lobsters (<90 mm) increased by 97% in the no-take zone (from 3 to 7 lobsters/line) and by 140% in the potting-only zones (argued by the authors to be due to spill-over effects; from 2 to 4–7 lobsters/line), where they appeared greater than in the fully fished areas where abundance remained constant (2–4 lobsters/line). The size of large lobsters (≥90 mm) increased by 5% inside the no-take zone between 2004 (98 mm) and 2007 (103 mm) and became 9% larger than in the potting-only zone and fished areas where lobster size decreased by 2–3% (from 98 to 95 mm). The size of small lobsters did not change over time and was similar across all areas. Abundance of velvet crabs *Necora puber* decreased by 65% inside the no-take zone over time (from 2 to 1 crab/line; argued by the authors to be due to increased predation by lobsters) and decreased even more in the potting-only zones (from 5–6 to 1 crab/line), and appeared lower than in the fully fished areas where it increased (from 0–6 to 1–7 crabs/line). The average size of velvet crabs did not change over time and was similar across all areas. Abundance of brown crabs *Cancer pagurus* did not change over time inside the no-take zone (0.3 crab/line), nor in the potting-only zone and fished areas (from 0.3–2 crabs/line). The average size of brown crabs increased by 25% inside the no-take zone between 2004 (115 mm) and 2007 (144 mm) and became greater than in the potting-only zones (123–128 mm) but not in fully fished areas (116–130 mm). Abundance of spider crabs *Maja squinado* was similar in 2004 and 2007 for all areas but varied spatially (with the no-take zone having lower abundance). The average size of spider crabs did not change over time and was similar across all areas. Lundy Island marine protected area was designated as a voluntary reserve in 1971 (statutory since 1986). In 2003, it included a 4 km^2 no-take zone (no fishing or harvesting allowed), the rest being a refuge zone only allowing crab and lobster potting. In 2004–2007, lobsters and crabs were surveyed inside the no-take zone, at two locations in the refuge zone, and two distant fished locations (20–100 km away) (2 sites/location). Four lines of standard commercial baited shellfish pots were deployed (10 pots/line) at each site for 24 h. Upon retrieval, lobsters and crabs were counted and measured (carapace length). The pots were redeployed for five consecutive days each year.

A replicated, randomized, controlled study in 2008–2012 of nine rocky reef sites inside a marine park in Shark Bay, Indian Ocean, southeastern New South Wales, Australia (8) found that five years after designating a marine park with various levels of fishing restrictions, the abundance of blacklip abalone *Haliotis rubra* was higher in sites with full fishing prohibition compared to those with partial prohibition and compared to sites outside the park (all fishing allowed). There were more abalone in sites with full fishing prohibition (4.3 individuals/transect) compared to sites with partial prohibition (0.9) and sites outside the park (1.9). In 2007, a marine park was established which included zones where all fishing was prohibited, and zones with partial prohibition (commercial fishing prohibited but recreational fishing and harvesting allowed). Twice annually between 2008 and 2012, samples were collected at nine randomly selected sites: three within each prohibition level inside the park, three outside the park where all fishing is allowed. Three 30 m transects/site were randomly deployed at 1–3 m depth, and abundance of blacklip abalone estimated from one 1 m strip/transect.

A site comparison study in 2010 of six sites in two zones inside a marine protected area in the Bristol Channel, UK (9 – similar set-up as 10) found that sites in the no-take zone (where all fishing had been prohibited for six years) had more and bigger European lobsters *Homarus gammarus* than sites outside in the refuge zone where potting was allowed. Lobsters were caught in higher abundance inside the no-take zone (514) than outside (152) and grew bigger inside (99 mm) than outside (86 mm). In addition, more lobsters were above the minimum landing size (90 mm) inside the no-take zone (75% of lobsters) than in the refuge zone (36% of lobsters). A higher proportion of egg-bearing females were found in the no-take zone (31%) compared to the refuge zone (7%). Overall, similar proportions of injured lobsters were found inside the no-take zone (33%) and inside the refuge zone (26%). The percentage of diseased lobsters was higher inside the no-take zone (27%) compared to the refuge zone (18%). Lundy Island marine protected area was designated as a voluntary reserve in 1971 (statutory since 1986). In 2003, it included a 4 km^2 no-take zone (no fishing or harvesting allowed), the rest being a refuge zone only allowing crab and lobster potting (all other fishing is prohibited). In 2010, six sites inside the protected area were surveyed: two within the no-take zone and four in the refuge zone. At each site, one line of 35 baited pots was deployed for 24–48 h, and all lobsters caught were measured (carapace length), sexed, assessed for injuries and diseases, and released back into the water. This process was repeated continuously over four days in May and again in June.

A site comparison study in 2011 of six sites in two zones inside a marine protected area in the Bristol Channel, UK (10 – similar set-up as 9) found that sites in the no-take zone (where all fishing had been prohibited for seven years) had more and bigger European lobsters *Homarus gammarus* than sites outside in the zone where potting was allowed. Lobsters were more abundant inside the no-take zone (40/line of 35 pots) than outside (20), and grew bigger inside (93 mm) than outside (85 mm). In addition, more lobsters were above the minimum landing size (90 mm) inside the no-take zone (61% of lobsters) than outside (32% of lobsters). Because more and bigger lobsters occurred inside the no-take zone, more were found injured (inside: 41%; outside: 19%; assumed to be likely due to increases in fighting behaviour). The percentage of diseased lobsters was similar inside (28%) and outside (17%) the no-take zone. Lundy Island marine protected area was designated as a voluntary reserve in 1971 (statutory since 1986). In 2003, it included a 4 km^2 no-take zone (no fishing or harvesting allowed), the rest being a refuge zone only allowing crab and lobster potting (all other fishing is prohibited). In August 2011, six sites inside the protected area were surveyed: two within the no-take zone and four outside in the refuge zone. At each site, one line of 35 baited pots was deployed for 24 h, and all lobsters caught were measured (carapace length) and assessed for injuries and diseases.

A site comparison study in 2011–2012 of two areas within a marine protected area in the Caribbean Sea, Mexico (11) found that Caribbean spiny lobsters *Panulirus argus* grew larger in an area where commercial fishing was banned compared to a fished area, and that the majority of the lobster population in the unfished area remained protected. Lobster sizes were greater in the unfished area (94 mm) compared to the fished area (73 mm). In the unfished area, this corresponded to 99% of lobsters being bigger than the minimum legal catch size (74.5 mm), while in the fished area it corresponded to only 25%. In addition, an estimated 20% of the lobster population occurring in the unfished area moved to the fished area, thus 80% remained protected. The study was carried out in a Biosphere Reserve (year of designation unspecified) which restricted commercial fishing to shallow depths (<20 m) and banned it where depths exceed 20 m (see paper for details). In August–September 2011, lobsters were hand-caught from the unfished area, tagged, sized (carapace length) and released in the unfished area (379 in total). During the 2011/2012 fishing season in the fished area, all lobsters caught by fishermen were sized, and tagged lobsters recorded. A tag-recapture model based on the number of recaptured tagged lobsters (20 in total) was used to estimate the percentage of the lobster population moving from the protected to the fished area.

A site comparison study in 2013 of four rocky seabed sites inside a marine park with a zonation system in the North Atlantic Ocean, southwest Portugal (12) found that sites prohibiting nearly all fishing had similar invertebrate species richness to sites where fishing was mostly allowed, two years after implementation. Sites prohibiting nearly all fishing had six species and sites where fishing was mostly allowed had seven species. In addition, abundances of specific groups appeared to vary between sites prohibiting nearly all fishing and sites mostly allowing fishing (sea urchins: 7 vs 31; brittle stars: 4 vs 63; starfish: 0–8 vs 1–39; sea cucumbers: 12 vs 31; octopus: 1 vs 5; data not statistically tested; unit unspecified). Fishery restrictions inside the park were implemented in 2011. In February–May 2013, four partially protected sites were sampled (0–15 m depth): two where nearly all professional and recreational fishing were prohibited (only barnacle extraction permitted), and two where fishing was mostly allowed (bottom trawling and recreational fishing not allowed on Wednesdays). Divers identified and counted all macro-invertebrates (size unspecified) along four 10 × 2 m transects/site.

A site comparison study in 2007–2013 of 11–23 coral reef sites inside Glover's Reef Marine Reserve, Caribbean Sea, Belize (13) found that the effects of a protected no-take area on the abundances and sizes of queen conch *Lobatus gigas* and Caribbean spiny lobster *Panulirus argus*, compared to the protected general-use zone with only some restrictions, varied with the size of individuals. Inside the marine reserve, 14 to 20 years after its designation, abundance of mature conch (>5 mm lip thickness) increased over time in the no-take sites (from 4/ha in 2007 to 17/ha in 2013), and was greater than in the general-use sites where the change (from 1 to 2/ha) was not significant. Immature conch (<5 mm) abundance increased similarly in no-take (from 4 to 53/ha) and general-use sites (from <1 to 33/ha). The lip thickness of mature conch decreased similarly over time at all sites (from 11 to 9 mm). The shell length of immature conch decreased similarly over time in no-take sites (from 221 to 182 mm) and general-use sites (from 234 to 186 mm). Abundance of legal-size (>76 mm carapace length) and sub-legal (<76 mm) lobsters increased over time in the no-take sites (legal-size: from 6 to 16/ha; sub-legal: from 1 to 3/ha) but did not change in the general-use sites (legal-size: non-significant change from 6 to 5/ha; sub-legal: remained at 6/ha). The size of all lobsters decreased over time in both no-take sites (legal-size: from 120 to 110 mm; sub-legal: from 59 to 52 mm) and general-use sites (legal-size: from 110 to 100 mm; sub-legal: from 59 to 52 mm). Glover's Reef Atoll was designated as a Marine Reserve in 1993 and included a no-take area (79.6 km^2) and a general-use area with fishery restrictions (including: ban on the use of SCUBA to collect any seafood, closed seasons, and size limits for queen conch and spiny lobster). Once a year in April–June 2007–2013, conch and lobsters were surveyed at 1.6 m average depth inside the no-take area (6–18 sites/year) and inside the general-use zone (5 sites/year). At each site (0.04–1.43 ha), snorkelers counted and measured all conch (shell length; lip thickness) and lobster (carapace length).

(1) Acosta C.A. (2002) Spatially explicit dispersal dynamics and equilibrium population sizes in marine harvest refuges. *ICES Journal of Marine Science,* 59, 458–468.

(2) Acosta C. & Robertson D. (2003) Comparative spatial ecology of fished spiny lobsters *Panulirus argus* and an unfished congener *P. guttatus* in an isolated marine reserve at Glover's Reef atoll, Belize. *Coral Reefs,* 22, 1–9.

(3) Ceccherelli G., Casu D., Pala D., Pinna S. & Sechi N. (2006) Evaluating the effects of protection on two benthic habitats at Tavolara-Punta Coda Cavallo MPA (North-East Sardinia, Italy). *Marine Environmental Research,* 61, 171–185.

(4) Gianguzza P., Chiantore M., Bonaviri C., Cattaneo-Vietti R., Vielmini I. & Riggio S. (2006) The effects of recreational *Paracentrotus lividus* fishing on distribution patterns of sea urchins at Ustica Island MPA (Western Mediterranean, Italy). *Fisheries Research,* 81, 37–44.

(5) Ceccherelli G. Pinna S. & Sechi N. (2009) Evaluating the effects of protection on *Paracentrotus lividus* distribution in two contrasting habitats. *Estuarine, Coastal and Shelf Science*, 81, 59–64.

(6) Sonnenholzner J.I. Ladah L.B. & Lafferty K.D. (2009) Cascading effects of fishing on Galapagos rocky reef communities: Reanalysis using corrected data. *Marine Ecology Progress Series,* 375, 209–218.

(7) Hoskin M.G., Coleman R.A., Von Carlshausen E. & Davis C.M. (2011) Variable population responses by large decapod crustaceans to the establishment of a temperate marine no-take zone. *Canadian Journal of Fisheries and Aquatic Sciences*, 68, 185–200.

(8) Wootton E.C., Woolmer A.P., Vogan C.L., Pope E.C., Hamilton K. M. & Rowley A.F. (2012) Increased disease calls for a cost-benefits review of marine reserves. *PLoS One*, 7, e51615.

(9) Coleman, M.A., Palmer-Brodie, A., & Kelaher, B.P. (2013) Conservation benefits of a network of marine reserves and partially protected areas. *Biological Conservation*, 167, 257–264.

(10) Davies C.E., Johnson A.F., Wootton E.C., Greenwood S.J., Clark K.F., Vogan C.L. & Rowley A.F. (2014) Effects of population density and body size on disease ecology of the European lobster in a temperate marine conservation zone. *ICES Journal of Marine Science*, 72, 128–138.

(11) Ley-Cooper K., De Lestang S., Phillips B.F. & Lozano-Álvarez E. (2014) An unfished area enhances a spiny lobster, *Panulirus argus*, fishery: implications for management and conservation within a Biosphere Reserve in the Mexican Caribbean. *Fisheries Management and Ecology*, 21, 264–274.
(12) Gil Fernández C., Paulo D., Serrão E.A. & Engelen A.H. (2016) Limited differences in fish and benthic communities and possible cascading effects inside and outside a protected marine area in Sagres (SW Portugal). *Marine Environmental Research,* 114, 12–23.
(13) Tewfik A., Babcock E.A., Gibson J., Perez V.R.B. & Strindberg S. (2017) Benefits of a replenishment zone revealed through trends in focal species at Glover's Atoll, Belize. *Marine Ecology Progress Series,* 580, 37–56.

11.9. Designate a Marine Protected Area and prohibit static fishing gear

- We found no studies that evaluated the effects of designating a Marine Protected Area and prohibiting static fishing gear on subtidal benthic invertebrate 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

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear entering in contact with the seabed (Collie *et al*. 2000). Static fishing gear such as pots and traps, although usually considered less impactful than mobile gears, can be locally damaging to the seabed and subtidal benthic invertebrates directly located under or in their vicinity. Specific areas can be designated as protected, and specific management measures taken to control for static gear (Blyth *et al*. 2002). Inside protected areas where static gear is prohibited, the threat from these practices to subtidal benthic invertebrates is removed, and previously impacted populations are, in theory, able to recover over time (Blyth *et al*. 2004). However, species and populations are still subjected to the effects of other fishing activities allowed (for instance recreational fishing).

Evidence related to similar intervention outside of a protected area are summarised under "Threat: Biological resource use – Cease or prohibit static fishing gears".

Blyth R.E., Kaiser M.J., Edwards-Jones G. & Hart P.J. B. (2004) Implications of a zoned fishery management system for marine benthic communities. *Journal of Applied Ecology*, 41, 951–961.
Blyth R.E., Kaiser M.J., Edwards-Jones G. & Hart P.J.B. (2002) Voluntary management in an inshore fishery has conservation benefits. *Environmental Conservation*, 29, 493–508.
Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

11.10. Designate a Marine Protected Area and limit the density of traps

- We found no studies that evaluated the effects of designating a Marine Protected Area and limiting the density of traps on subtidal benthic invertebrate 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

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear entering in contact with the seabed (Collie *et al*. 2000). Traps or pots are often used to fish for crabs or lobsters and consist of structures into which species of commercial interest enter through funnels which encourage entry, but limit escape. Specific areas can be designated as protected, in which the density of traps is limited (Acheson 1998; Miller 1976). Inside protected areas where the density of traps is limited, the threat from these practices to subtidal benthic invertebrates is removed, and previously impacted populations are, in theory, able to recover over time. However, species and populations are still subjected to the effects of other fishing activities allowed (for instance mobile fishing gears).

Evidence related to similar intervention outside of a protected area are summarised under "Threat: Biological resource use – Limit the density of traps".

Acheson J. (1998) Lobster trap limits: A solution to a communal action problem. *Human Organization*, 57, 43–52.

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

Miller R.J. (1976) North American crab fisheries: regulations and their rationales. *Fishery Bulletin*, 74, 623–633.

11.11. Designate a Marine Protected Area and only allow hook and line fishing

- **One study** examined the effects of allowing only hook and line fishing in marine protected areas on subtidal benthic invertebrate populations. The study was in the Skagerrak[1] (Norway).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Crustacean abundance (1 study):** One replicated, controlled, before-and-after study in the Skagerrak[1] found that sites inside a protected area only allowing hook and line fishing had greater increases in lobster abundance over the four years after the area was designated compared to unprotected fully fished sites.
- **Crustacean condition (1 study):** One replicated, controlled, before-and-after study in the Skagerrak[1] found that sites inside a protected area only allowing hook and line fishing had greater increases in lobster size over the four years after the area was designated compared to unprotected fully fished sites.

Background

'Hook and line' fishing is a term used for a range of fishing methods that use short fishing lines with hooks. Hook and line fishing is more selective than other types of fishing and has little impact on the seabed. In addition, bycatch species can often be returned to the sea alive because the lines are only in place for a short time. These methods also reduce the direct contact with the seabed, any unintentional physical harm and disturbances, and reduce the amount of bycatch. Specific areas can be designated as protected, and specific management measures taken to control for fishing gears. Inside protected areas where only hook and line fishing is allowed, the threat from these practices to subtidal benthic invertebrates is removed, and previously impacted populations are, in theory, able to recover over time.

When this intervention occurred outside of a protected area, evidence has been summarised under "Threat: Biological resource use – Use hook and line fishing instead of other fishing methods".

Moland E., Olsen E.M., Knutsen H., Garrigou P., Espeland S.H., Kleiven A.R., André C. & Knutsen J.A. (2013) Lobster and cod benefit from small-scale northern marine protected areas: inference from an empirical before–after control-impact study. *Proceedings of the Royal Society B: Biological Sciences,* 280, 1754.

A replicated, controlled, before-and-after study in 2006–2010 in six areas of seabed off the Norwegian Skagerrak coast (1) found that, during the four years after being designated, protected areas only allowing hook and line fishing had greater increases in the number and size of European lobster *Homarus gammarus*, compared to fully fished areas. Before designation, lobster abundance (as catch/unit effort) was typically similar in all areas (protected: 0.5 lobster/trap; fully fished areas: 0.5–1.5 lobsters/trap). Over time, abundance increased at all sites, but increased more in protected areas, and after four years had increased by 245% in protected areas, (1–3 lobsters/trap), but only by 87% in fully fished areas (0.5–2.5 lobsters/trap). Before designation, lobster size was similar across areas (protected: 23–24 cm; fully fished: 24–25 cm). Over time, size increased at all sites, but more in the protected areas, and after four years had increased by 12–15% (26–28 cm), but only by 3% in fully fished areas (24–25 cm). In September 2006, three marine protected areas only allowing hook and line fishing were established. Annually in 2006–2010, lobsters were sampled inside each protected area and at three fully fished areas (no gear restriction; one adjacent to each protected area) using traps (25/area) deployed at 10–30 m depth. After 24 h, all lobsters in traps were counted and measured (carapace length). Traps were redeployed daily over four days.

(1) Moland E., Olsen E.M., Knutsen H., Garrigou P., Espeland S.H., Kleiven A.R., André C. & Knutsen J.A. (2013) Lobster and cod benefit from small-scale northern marine protected areas: inference from an empirical before–after control-impact study. *Proceedings of the Royal Society B: Biological Sciences,* 280, 1754.

11.12. Designate a Marine Protected Area and limit the number of fishing vessels

- We found no studies that evaluated the effects of designating a Marine Protected Area and limiting the number of fishing vessels on subtidal benthic invertebrate 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

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear entering in contact with the seabed (Collie *et al*. 2000). Specific areas can be designated as protected, and specific management measures taken to limit the number of fishing vessels allowed. This could potentially reduce fishing effort in the protected area, thereby reducing the impact on the seabed, the amount of unwanted catch, and overall threat to subtidal benthic invertebrates. However, species and populations are still subjected to the effects of other allowed activities (for instance hand harvest or recreational boating).

Evidence related to similar intervention outside of a protected area are summarised under "Threat: Biological resource use – Limit the number of fishing vessels".

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

11.13. Designate a Marine Protected Area and set a no-anchoring zone

- We found no studies that evaluated the effects of designating a Marine Protected Area and setting a no-anchoring zone on subtidal benthic invertebrate 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

Anchoring of boats (and other vessels) can impact subtidal benthic invertebrates through physical damage from anchors and chains (Griffith *et al*. 2017). Structurally complex seabed habitats, such as seagrass and mussel beds, or oyster and coral reefs, are considered particularly at risk from recreational anchoring (Hammerstrom *et al.* 2007). Specific areas can be designated as protected, and specific management measures taken to control for impactful activities such as anchoring (Axelson *et al*. 2012). Setting a no-anchoring zone(s) in marine protected areas can help reduce anchoring-related pressures on subtidal benthic invertebrates, potentially allowing them to naturally recover over time. However, species and populations are still subjected to the effects of other allowed activities.

Evidence related to similar intervention outside of a protected area are summarised under "Threat: Transportation and service corridors – Set limits or reduce the area where ships can anchor".

Axelsson M., Allen C. & Dewey S. (2012) *Survey and monitoring of seagrass beds at Studland Bay, Dorset – second seagrass monitoring report*. Seastar Survey Ltd, Report to The Crown Estate and Natural England.
Griffiths C.A., Langmead O.A., Readman J.A.J. & Tillin H.M. (2017) *Anchoring and Mooring Impacts in English and Welsh Marine Protected Areas: Reviewing sensitivity, activity, risk and management.* A report to Defra Impacts Evidence Group.
Hammerstrom K.K., Kenworthy W.J., Whitfield P.E. & Merello M.F. (2007) Response and recovery dynamics of seagrasses *Thalassia testudinum* and *Syringodium filiforme* and macroalgae in experimental motor vessel disturbances. *Marine Ecology Progress Series,* 345, 83–92.

11.14. Designate a Marine Protected Area and prohibit the harvest of scallops

- We found no studies that evaluated the effects of designating a Marine Protected Area and prohibiting the harvest of scallops on subtidal benthic invertebrate 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

Commercial (and often recreational) harvest of scallops is usually undertaken using dredges, and as such can impact subtidal benthic invertebrates through removal of untargeted species and damage to the seabed (Bull 1989). Specific areas can be designated as protected, and specific management measures taken to cease or prohibit the commercial and/or recreational harvest of scallops (Mangi *et al.* 2011). Inside protected areas where this activity is prohibited, the threat from scallop harvesting to scallop populations and associated benthic communities is removed, and previously impacted populations are, in theory, able to recover over time. However, species and populations are still subjected to the effects of other activities allowed.

Evidence related to similar intervention outside of a protected area are summarised under "Species management – Cease or prohibit harvest of scallops", and "Threat: Biological resource use – Cease or prohibit dredging".

Bull M.F. (1989) *The New Zealand scallop fishery: a brief review of the fishery and its management*. Edited by: MLC Dredge, WF Zacharin and LM Joli, 42.

Mangi S.C., Rodwell L.D. & Hattam C. (2011) Assessing the impacts of establishing MPAs on fishermen and fish merchants: the case of Lyme Bay, UK. *Ambio*, 40, 457.

11.15. Designate a Marine Protected Area and prohibit the harvest of conch

- **One study** examined the effects of prohibiting the harvest of conch in marine protected areas on their populations and/or other subtidal benthic invertebrates. The study was in the North Atlantic Ocean[1] (British Overseas Territories).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Mollusc abundance (1 study):** One site comparison study in the North Atlantic Ocean[1] found that a marine protected area prohibiting the commercial harvest of conch had more conch after five years compared to a fished area.
- **Mollusc condition (1 study):** One site comparison study in the North Atlantic Ocean[1] found that a marine protected area prohibiting the commercial harvest of conch had smaller adult conch after five years compared to a fished area.

Background

Conch populations have significantly declined due to overharvesting for commercial and recreational purposes (Theile 2001). Specific areas can be designated as protected, and specific management measures taken to cease or prohibit the harvest of conch (Béné & Tewfik 2003; Stoner *et al*. 2012). Inside protected areas where this activity is prohibited, the threat from conch harvesting to conch populations and associated benthic communities is removed, and previously impacted populations are, in theory, able to recover over time (Stoner *et al.* 2012). When this intervention occurred outside of a marine protected area, evidence for the effects on conch populations is summarised under "Species management – Cease or prohibit the harvest of conch". Evidence for related interventions is summarised under "Threat: Biological resource use".

Béné C. & Tewfik A. (2003) Biological evaluation of marine protected area: evidence of crowding effect on a protected population of queen conch in the Caribbean. *Marine Ecology*, 24, 45–58.

Stoner A.W., Davis M.H. & Booker C.J. (2012) Negative consequences of Allee effect are compounded by fishing pressure: comparison of queen conch reproduction in fishing grounds and a marine protected area. *Bulletin of Marine Science*, 88, 89–104.

Theile S. (2001) *Queen conch fisheries and their management in the Caribbean*. Brussels: TRAFFIC Europe.

A site comparison study in 1998 in areas of algal seabed, sandy seabed, or seagrass bed in the North Atlantic Ocean, Turks and Caicos Islands, British Overseas Territories (1) found that inside a marine protected area that had been prohibiting the commercial harvest of conch for five years, abundance of queen conch *Strombus gigas* was higher compared to a fished area, but effects varied with the age of conch and habitat type. Total conch abundance (juveniles and adults) was higher in the closed (555 conch/ha) compared to the fished area (277 conch/ha). Abundance of adult conch (≥4 mm lip thickness) was higher in the closed compared to the fished area for algal habitat (closed: 833 vs fished: 86) and sandy habitat (78 vs 28), but not statistically different for seagrass habitat (410 vs 24). Abundance of juvenile conch (<4 mm lip thickness) was similar inside and outside the closed area for all habitats (179–483 vs 85–497). In addition, adult conch were smaller in the closed area compared to the fished area (186 vs 204 mm shell length). In 1993 a conch sanctuary (approximately 17.5 km^2) prohibiting the commercial harvest of conch was experimentally established. In June–November 1998 at an unspecified number of sites both inside and outside the sanctuary, divers counted and measured (shell length and lip thickness) all conch in 6 x 60 m transects at 0.2–12 m depth.

(1) Béné C. & Tewfik A. (2003) Biological evaluation of marine protected area: evidence of crowding effect on a protected population of queen conch in the Caribbean. *Marine Ecology*, 24, 45–58.

11.16. Designate a Marine Protected Area and prohibit the harvest of sea urchins

- **Two studies** examined the effects of prohibiting the harvest of sea urchins in marine protected areas on their populations and/or other subtidal benthic invertebrates. The studies were in the North Pacific Ocean[1,2] (USA).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Echinoderm abundance (1 study):** One replicated, site comparison study in the North Pacific Ocean[2] found that marine protected areas prohibiting the harvest of red sea urchins had higher adult sea urchin biomass six to 33 years after their designations, compared to harvested areas.
- **Echinoderm reproductive success (1 study):** One replicated, site comparison study in the North Pacific Ocean[2] found that marine protected areas prohibiting the harvest of red sea urchins had higher urchin population reproductive biomasses, but similar reproductive indices six to 33 years after their designations, compared to harvested areas.
- **Echinoderm condition (1 study):** One replicated, site comparison study in the North Pacific Ocean[2] found that marine protected areas prohibiting the harvest of red sea urchins had bigger adult sea urchins six to 33 years after their designations, compared to harvested areas.
- **Mollusc abundance (1 study):** One replicated, site comparison study in the North Pacific Ocean[1] found that marine protected areas prohibiting the harvest of red sea urchins (year of designation unspecified) had more juvenile red abalone and juvenile flat abalone compared to harvested areas, and that juvenile abalone abundance was positively related to sea urchin abundance.

Background

Sea urchins can represent key species within a marine system, with other species crucially depending on their presence to thrive (Coyer *et al.* 1993; Day & Branch 2002) and to retain balance in the ecosystem (Blamey *et al.* 2014). Commercial, but also recreational, harvest of edible sea urchins has led to significant ecological changes, not only for sea urchin populations, but also for other species suffering from secondary negative consequences (i.e. a ripple effect; for instance, urchin harvest can negatively affect protected species of abalone; Rogers-Bennett & Pearse 2001). Specific areas can be designated as protected, and specific management measures taken to prohibit the harvest of sea urchins (Béné & Tewfik 2003; Stoner *et al*. 2012).

Inside protected areas where this activity is prohibited, the threat from sea urchin harvesting to urchin populations and associated communities is removed, and previously impacted populations are, in theory, able to recover over time (Stoner *et al.* 2012).

When this intervention occurred outside of a marine protected area, evidence for the effects on sea urchin populations is summarised under "Species management – Cease or prohibit the harvest of sea urchin". Evidence for related interventions is summarised under "Threat: Biological resource use".

Béné C. & Tewfik A. (2003) Biological evaluation of marine protected area: evidence of crowding effect on a protected population of queen conch in the Caribbean. Marine Ecology, 24, 45–58.

Blamey L.K., Plagányi É.E. & Branch G.M. (2014) Was overfishing of predatory fish responsible for a lobster-induced regime shift in the Benguela? *Ecological Modelling*, 273, 140–150.

Coyer J.A., Ambrose R.F., Engle J.M. & Carroll J.C. (1993) Interactions between corals and algae on a temperate zone rocky reef: mediation by sea urchins. *Journal of Experimental Marine Biology and Ecology*, 167, 21–37.

Ceccherelli G., Pinna S. & Sechi N. (2009) Evaluating the effects of protection on *Paracentrotus lividus* distribution in two contrasting habitats. *Estuarine, Coastal and Shelf Science*, 81, 59–64.

Day E. & Branch G.M. (2000) Relationships between recruits of abalone *Haliotis midae*, encrusting corallines and the sea urchin *Parechinus angulosus*. *South African Journal of Marine Science*, 22, 137–144.

Rogers-Bennett L. & Pearse J.S. (2001) Indirect benefits of marine protected areas for juvenile abalone. *Conservation Biology*, 15, 642–647.

Stoner A.W., Davis M.H. & Booker C.J. (2012) Negative consequences of Allee effect are compounded by fishing pressure: comparison of queen conch reproduction in fishing grounds and a marine protected area. Bulletin of Marine Science, 88, 89–104.

A replicated, site comparison study in 1996–1997 of six rocky seabed sites off the coast of central and northern California, North Pacific Ocean, USA (1) found that marine protected areas prohibiting the commercial harvest of red sea urchins *Strongylocentrotus franciscanus* had higher abundances of juvenile red abalone *Haliotis rufescens* and juvenile flat abalone *Haliotis walallensis* compared to areas where commercial harvesting occurred. Abundances of both species were higher in protected areas (red abalone: 8–139/plot; flat abalone: 0–18/plot) compared to harvested areas (red abalone: 0–39/plot; flat abalone: 0–9). In addition, juvenile abalone abundance was significantly positively related to sea urchin abundance, and inside protected areas 33% of juvenile abalone were found protected under sea urchin spine canopies. In October 1996 and August 1997, three marine protected areas (year of designation unspecified) prohibiting the commercial harvest of red sea urchins and three areas where urchin harvest occurred were surveyed. Juvenile red and flat abalone were counted in 24 x 30 m plots/site (5–8 m depth).

A replicated, site comparison study in 2009–2011 of 30 sites around the Northern Channel Islands, southern California, North Pacific Ocean, USA (2) found that, six to 33 years after their designations, marine protected areas prohibiting the harvest of red sea urchin *Mesocentrotus franciscanus* had bigger adult urchins, higher adult total biomass and reproductive biomass, but similar urchin reproductive indices (ratio of reproductive to total biomasses), compared to sites where urchin harvest was allowed. Adult urchins diameter was 6% bigger inside the marine protected areas compared to outside. Adult total biomass was 16%, and reproductive biomass was 23% greater inside the marine protected areas compared to outside. Once a year in summer between 2009 and 2011, eleven sites within seven marine protected areas and 13 sites outside of marine protected areas were surveyed at 6 m and 13 m depths (143 surveys in total). One marine protected area was designated in 1978, and six in 2003. Despite having different levels of activity restrictions, all areas prohibited the harvest of the red sea urchin. Divers counted all urchins >25 mm test diameter along two 60 m^2 transects/site/water depth. Fifteen to 20 urchins >50 mm (test diameter) were collected, measured, and their flesh and reproductive glands weighed. For each are. adult total biomass (using total urchin weight) and reproductive biomass (using urchins reproductive gland weight) were calculated from urchins count and weight data.

(1) Rogers-Bennett L. & Pearse J.S. (2001) Indirect benefits of marine protected areas for juvenile abalone. *Conservation Biology*, 15, 642–647.
(2) Teck S.J., Lorda J., Shears N.T., Bell T.W., Cornejo-Donoso J., Caselle J.E., Hamilton S.L. & Gaines S.D., (2017) Disentangling the effects of fishing and environmental forcing on demographic variation in an exploited species. *Biological Conservation*, 209, 488–498.

11.17. Designate a Marine Protected Area and introduce some fishing restrictions (types unspecified)

- **Four studies** examined the effects of introducing unspecified types of fishing restrictions in marine protected areas on subtidal benthic invertebrate populations. Two studies were in the Indian Ocean[1,2] (Seychelles), one was a global systematic review[3], and one was in the Mediterranean Sea[4] (Italy).

COMMUNITY RESPONSE (2 STUDIES)

- **Overall community composition (2 studies):** One of two site comparison studies (one replicated) in the Indian Ocean[1] and the Mediterranean Sea[4] found that a marine protected area with unspecified fishing restrictions (year of designation unspecified) had a different combined invertebrate and algae community composition[1], while the other[4] (time since designation unspecified) found similar compositions compared to fished areas.

POPULATION RESPONSE (3 STUDIES)

- **Overall abundance (1 study):** One replicated, site comparison study in the Mediterranean Sea[4] found that a marine protected area with unspecified fishing restrictions had similar invertebrate abundance compared to unprotected fished areas (time since designation unspecified).
- **Bryozoan abundance (1 study):** One site comparison study in the Indian Ocean[1] found that a marine protected area with unspecified fishing restrictions (year of designation unspecified) had similar abundance of bryozoans compared to fished areas.
- **Crustacean abundance (1 study):** One global systematic review[3] found that marine protected areas with unspecified fishing restrictions had more lobsters compared to fished areas.
- **Echinoderm abundance (2 studies):** One of two site comparison studies (one replicated) in the Indian Ocean[1,2] found that marine protected areas with unspecified fishing restrictions had more sea cucumbers after more than 20 years[2], but the other found fewer sea lilies (year of designation unspecified)[1], compared to fished areas.

- **Hydrozoan abundance (1 study):** One site comparison study in the Indian Ocean[1] found that a marine protected area with unspecified fishing restrictions (year of designation unspecified) had more hydrozoans compared to fished areas.
- **Mollusc abundance (1 study):** One global systematic review[3] found that marine protected areas with unspecified fishing restrictions had more scallops compared to fished areas.
- **Sponge abundance (1 study):** One site comparison study in the Indian Ocean[1] found that a marine protected area with unspecified fishing restrictions (year of designation unspecified) had more sponges compared to fished areas.
- **Tunicate abundance (1 study):** One site comparison study in the Indian Ocean[1] found that a marine protected area closed to fishing with unspecified fishing restrictions (year of designation unspecified) had similar abundance of ascidians/sea squirts (tunicates) compared to fished areas.

Background

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from gear entering in contact with the seabed (Collie *et al*. 2000). Specific areas can be designated as protected, and specific management measures taken to control for impactful activities, such as commercial, recreational, or artisanal fishing (Villamor & Becerro 2012), for instance by restricting specific gear or practices. Inside protected areas where some levels of fishing are prohibited, the threat to subtidal benthic invertebrates is removed, and previously impacted populations are, in theory, able to recover over time (Ley-Cooper *et al.* 2014). However, species and populations are still subjected to the effects of other fishing activities allowed (for instance recreational fishing).

Here, we present evidence for marine protected areas where the exact level or nature of the fishing restrictions are unclear or unspecified. Evidence for related interventions regarding fishing restrictions within protected areas is summarised under "Habitat protection".

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

Ley-Cooper K., De Lestang S., Phillips B.F. & Lozano-Álvarez E. (2014) An unfished area enhances a spiny lobster, *Panulirus argus*, fishery: implications for management and conservation within a Biosphere Reserve in the Mexican Caribbean. *Fisheries Management and Ecology*, 21, 264–274.

Villamor A. & Becerro M.A. (2012) Species, trophic, and functional diversity in marine protected and non-protected areas. *Journal of Sea Research*, 73, 109–116.

A site comparison study in 2001–2004 in areas of seabed in the Indian Ocean, off the south coast of South Africa (1) found that sites inside a marine protected area closed to fishing (exact restrictions unspecified) had a different overall invertebrate and algae community composition and abundances of three of five species groups compared to adjacent fished sites. Community data were presented as graphical analyses. Protected sites had statistically higher abundance (as percentage cover) of sponges (25%) and hydrozoans (9%) compared to fished sites (sponges: 19%; hydrozoans: 7%), lower abundance of sea lilies (closed: 6% vs fished: 10%), and similar abundances of sea quirts (15% vs 13%) and bryozoans (20% vs 24%) than fished sites. Annually in 2001–2004, video footage was recorded at 10–30 m depth at 2–7 sites surveyed inside the protected area (year of designation unspecified), and 4–13 sites outside. At each site, a 225 m^2 area was video-recorded. Footage was analysed and cover of five invertebrate taxa and algae assessed.

A replicated, site comparison study in 2008 of 21 sites in seven coral reefs areas across the inner islands of the Seychelles, Indian Ocean (2) found that sea cucumbers (thirteen species combined) tended to be more abundant inside marine protected areas prohibiting some fishing (exact restrictions unspecified) compared to adjacent fished areas. Seventy-six percent of all sea cucumbers (thirteen species combined) were found within protected areas. The average abundance of sea cucumbers appeared higher in protected areas (2/154 m^2), compared to fished areas (0/154 m^2), although no statistical test was reported. The probability of finding sea cucumbers was reported to be higher in protected areas (79%), compared to fished areas (48%). In April, divers counted sea cucumbers in three protected areas (established >20 years prior; date unspecified) and four unprotected areas (three sites/area) within sixteen 154 m^2 circles/site.

A systematic review of 27 studies published before February 2011 of marine protected areas partially prohibiting fishing (restrictions unspecified) across the world (3) found that they had greater abundances of scallops and lobsters compared to outside where fishing was fully allowed. Average lobster abundance was 0.53 times higher, and scallop density 2.33 times higher, inside marine protected areas compared to outside. Exact species were not specified. Abundance data were not reported, but the outcome of analysis was reported as statistical model results. The selected studies compared invertebrate abundance inside and outside 25 marine protected areas with partial fishing prohibition. The abundance data were extracted and used in a meta-analysis.

A replicated, site comparison study (year unspecified) of 28 sites across 14 rocky reef areas in the western Mediterranean Sea, Italy (4) found that protected areas with 'low human pressures' (restrictions unspecified) had similar overall invertebrate and algae community composition to unprotected areas with 'high human pressures', and similar invertebrate abundance. Community composition data were presented as graphical analyses. Percent cover of invertebrates was similar in protected (6.2%) and unprotected areas (3.7%). Invertebrates and algae were surveyed at two sites inside each of seven marine protected areas (fishing restrictions unspecified) and seven unprotected areas. All protected areas were established between 1997 and 1999 and reported to "preserve reefs from all human activities". At 30–40 m depth, 10 plots (0.2 m^2) were photographed at three 10 m^2 locations/site. Invertebrates and algae species were identified and their % cover estimated from each photograph. Date of study unspecified.

(1) Götz A., Kerwath S.E., Attwood C.G. & Sauer W.H.H. (2009) Effects of fishing on a temperate reef community in South Africa 2: Benthic invertebrates and algae. *African Journal of Marine Science*, 31, 253–262.

(2) Cariglia N., Wilson S.K., Graham N.A.J., Fisher R., Robinson J., Aumeeruddy R., Quatre R. & Polunin N.V.C. (2013) Sea cucumbers in the Seychelles: effects of marine protected areas on high-value species. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 23, 418–428.

(3) Sciberras M., Jenkins S.R., Kaiser M.J., Hawkins S.J. & Pullin A.S. (2013) Evaluating the biological effectiveness of fully and partially protected marine areas. *Environmental Evidence*, 2, 4.

(4) Piazzi L., La Manna G., Cecchi E., Serena F. & Ceccherelli G. (2016) Protection changes the relevancy of scales of variability in coralligenous assemblages. *Estuarine, Coastal and Shelf Science*, 175, 62–69.

11.18. Designate a Marine Protected Area and prohibit aquaculture activity

• **One study** examined the effects of prohibiting aquaculture activity in a protected area on subtidal benthic invertebrate populations. The study was in Tapong Bay lagoon[1] (Taiwan).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Crustacean abundance (1 study):** One before-and-after study in Tapong Bay lagoon[1] found that two and a half years after removing oyster aquaculture in a marine protected area, the biomasses of amphipods and shrimps had decreased, and that the biomass of crabs had not changed.
- **Mollusc abundance (1 study):** One before-and-after study in Tapong Bay lagoon[1] found that two and a half years after removing oyster aquaculture in a marine protected area, the biomasses of gastropods and bivalves had decreased.
- **Worm abundance (1 study):** One before-and-after study in Tapong Bay lagoon[1] found that two and a half years after removing oyster aquaculture in a marine protected area, the biomass of polychaete worms had stayed the same.

Background

Aquaculture systems can negatively impact subtidal benthic invertebrate communities through pollution and diminished water quality (Wu *et al.* 1994). Ceasing or prohibiting aquaculture activity in an area, for instance following relocation to a different area or following decommissioning, would remove the source of harm and potentially allow for subtidal benthic invertebrate communities to recover over time (Johannessen *et al*. 1994). Specific areas can be designated as protected, and specific management measures taken to prohibit aquaculture (Lin *et al*. 2009). Inside protected areas where this activity is prohibited, the threat to subtidal invertebrate communities is removed, and previously impacted populations are, in theory, able to recover over time.

When this intervention occurred outside of a marine protected area, evidence has been summarised under "Threat: Pollution – Cease or prohibit aquaculture activity".

Johannessen P., Botnen H. & Tvedten Ø.F. (1994) Macrobenthos: before, during and after a fish farm. *Aquaculture Research,* 25, 55–66

Lin H.J., Shao K.T., Hsieh H.L., Lo W.T. & Dai X.X. (2009) The effects of system-scale removal of oyster-culture racks from Tapong Bay, southwestern Taiwan: model exploration and comparison with field observations. *ICES Journal of Marine Science*, 66, 797–810.

Wu R.S.S., Lam K.S., MacKay D.W., Lau T.C. & Yam V. (1994) Impact of marine fish farming on water quality and bottom sediment: A case study in the sub-tropical environment. *Marine Environmental Research,* 38, 115–145.

A before-and-after study in 1999–2004 of 39 sampling stations in Tapong Bay lagoon, southwestern Taiwan (1) found that removing oyster aquaculture in a marine protected area led to decreases in the biomasses of four out of six invertebrate groups, after two and a half years. Biomass of sea snails (gastropod molluscs) declined by 98% (before: 4.40; after: 0.06 g/m^2), bivalve molluscs by 97% (before: 274; after: 8.56 g/m^2), amphipods (crustaceans) by 98% (before: 0.51; after: 0.01 g/m^2), and shrimps by 50% (before: 0.12; after: 0.06 g/m^2). There were no significant changes in the biomasses of polychaete worms (before: 0.32; after: 1.55 g/m^2), and crabs (before: 1.59; after: 0.93 g/m^2). In 1997, Tapong Bay became a National Scenic Area and oyster culture, which was intensive in the area, was prohibited. In June 2002, all oyster racks were removed. Invertebrates (>0.5 mm) in the sediment were surveyed using a core (10 cm diameter; 20 cm depth) at 30 stations (3 cores/station) in August 1999, October 2002, and January and November 2004. Crabs and shrimps were sampled in 2001–2004 (unspecified number of surveys) using a net at nine stations (4 nets/station). All invertebrates were identified and wet-weighed.

(1) Lin H.J., Shao K.T., Hsieh H.L., Lo W.T. & Dai X.X. (2009) The effects of system-scale removal of oyster-culture racks from Tapong Bay, southwestern Taiwan: model exploration and comparison with field observations. *ICES Journal of Marine Science*, 66, 797–810.

11.19. Designate a Marine Protected Area without setting management measures, usage restrictions, or enforcement

• We found no studies that evaluated the effects of designating a Marine Protected Area without setting management measures, usage restrictions, or enforcement on subtidal benthic invertebrate 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

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear entering in contact with the seabed (Collie *et al*. 2000). Specific areas can be designated as protected, but often these are set without any clear management measures or usage restrictions in place, or enough enforcement (Guidetti *et al.* 2008). Inside such protected areas, it is unclear which activities and pressures are taking place, and if subtidal benthic invertebrates are able to naturally recover. While such areas, often referred to as "paper parks" (Rife *et al.* 2013), do exist, they are not recommended by the scientific community, and it is advised to always set clear management plans, objectives, and enforcement for marine protected areas (Di Minin & Toivonen 2015; Jones & De Santo 2016; Rife *et al.* 2013).

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

Di Minin E. & Toivonen T. (2015) Global protected area expansion: creating more than paper parks. *BioScience*, 65, 637–638.

Guidetti P., Milazzo M., Bussotti S., Molinari A., Murenu M., Pais A., Spano N., Balzano R., Agardy T., Boero F. & Carrada G. (2008) Italian marine reserve effectiveness: does enforcement matter? *Biological Conservation*, 141, 699–709.

Jones P.J. & De Santo E.M. (2016) Viewpoint–Is the race for remote, very large marine protected areas (VLMPAs) taking us down the wrong track? *Marine Policy*, 73, 231–234.

Rife A.N., Erisman B., Sanchez A. & Aburto-Oropeza O. (2013) When good intentions are not enough… Insights on networks of "paper park" marine protected areas. *Conservation Letters*, 6, 200–212.

11.20. Establish community-based fisheries management

• **One study** examined the effects of establishing community-based fisheries management on subtidal benthic invertebrate populations. The study was in the Foveaux Straight[1] (New Zealand).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Mollusc abundance (1 study):** One replicated, site comparison study in the Foveaux Straight[1] found that a customary fisheries area where management was community-based had more New

Zealand scallops compared to a protected area prohibiting all fishing and an area allowing recreational harvest.

- **Mollusc condition (1 study):** One replicated, site comparison study in the Foveaux Straight[1] found that a customary fisheries area where management was community-based, tended to have smaller New Zealand scallops compared to a protected area prohibiting all fishing and an area allowing recreational harvest.

Background

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear coming into contact with the seabed (Collie *et al*. 2000). Community-based fisheries management, sometimes referred to as indigenous customary management (Twist *et al*. 2016), assigns the management of marine resources to the local community who often possesses traditional and local knowledge crucial to local management. Community-based fisheries management is often based on a partial protection strategy, which uses one or more spatial management measures (for instance measures that restrict some aspect of the fishery; Cinner & Aswani 2007). Community-based management gives local communities the power to set regional fisheries bylaws and/or regulations and has been implemented into legislation in several countries (Ruddle 1998). By regulating and limiting fishing effort, and protecting the marine environment, community-based fisheries management can, in theory, reduce the impacts on the seabed, the amount of bycatch, and overall threat to subtidal benthic invertebrates (Twist *et al*. 2016).

Evidence for related intervention is summarised under "Threat: Biological resource use – Establish territorial user rights for fisheries".

Cinner J.E. & Aswani S. (2007) Integrating customary management into marine conservation. *Biological Conservation*, 140, 201–216.

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

Ruddle K. (1998) The context of policy design for existing communitybased fisheries management systems in the Pacific Islands. *Ocean and Coastal Management*, 40, 105–126.

Twist B.A., Hepburn C.D. & Rayment W.J. (2016) Distribution of the New Zealand scallop (*Pecten novaezealandiae*) within and surrounding a customary fisheries area. *ICES Journal of Marine Science*, 73, 384–393.

A replicated, site comparison study in 2013 of 20 sites in the Paterson Inlet, Foveaux Straight, New Zealand (1) found that sites within a customary fisheries area where management was community-based had more New Zealand scallops *Pecten novaezealandiae,* but they tended to be smaller, compared to adjacent sites in a marine protected area prohibiting all fishing (no-take reserve) and a recreational harvest-only area. Scallop abundance was higher inside the customary fisheries area (3.62 scallops/m^2) compared to the other sites (no-take: 0.63 scallops/m^2; recreational: 0.56 scallops/m^2). Scallops tended to be smaller in the customary fisheries area (104 mm), compared to the no-take reserve (110 mm), and the recreational area (132 mm; size data were not statistically tested). In June 2013, divers counted and measured scallops in three to nine transects (100 m^2) at each of 20 sites: six in the customary fisheries area (community-based management, see paper for details), three in the no-take reserve (designated in 2004), and three in the recreational harvest-only area.

(1) Twist B.A., Hepburn C.D. & Rayment W.J. (2016) Distribution of the New Zealand scallop (*Pecten novaezealandiae*) within and surrounding a customary fisheries area. *ICES Journal of Marine Science*, 73, 384–393.

11.21. Engage with stakeholders when designing Marine Protected Areas

- We found no studies that evaluated the effects of engaging with stakeholders when designating a Marine Protected Area on subtidal benthic invertebrate 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

Fishing can impact subtidal benthic invertebrates through species removal or habitat damage from fishing gear entering in contact with the seabed (Collie *et al.* 2000). Specific areas can be designated as protected, and specific management measures taken to control for impactful activities (Kelleher 1999). Engaging with stakeholders when designing protected or closed areas may empower resource users and lead to greater uptake, as well as minimising the social and economic effects of closing fishing grounds. This in turn can ensure protection enforcement and potentially the natural recovery of subtidal benthic invertebrates (Gleason *et al.* 2013; Pomeroy & Douvere 2008).

Related evidence is summarised under "Habitat protection – Establish community-based fisheries management".

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

Gleason M., Feller E.M., Merrifield M., Copps S., Fujita R.O.D., Bell M., Rienecke S. & Cook C. (2013) A transactional and collaborative approach to reducing effects of bottom trawling. *Conservation Biology,* 27, 470–479.

Kelleher G. (1999) Guidelines for marine protected areas. IUCN, Gland, Switzerland and Cambridge, UK.

Pomeroy R. & Douvere F. (2008) The engagement of stakeholders in the marine spatial planning process. *Marine Policy*, 32, 816–822.

12. Habitat restoration and creation

Background

Habitat destruction is the largest single threat to biodiversity and habitat fragmentation and degradation often reduces the quality of remaining habitat (Brooks *et al.* 2002). While habitat protection remains one of the most important and frequently used conservation intervention (see chapter on "Habitat protection"), in many parts of the world restoring damaged or lost habitats or, creating new habitat patches, may also be possible and benefit marine biodiversity (Dodson *et al.* 1997).

This chapter describes interventions that can be used to increase the diversity, health, and size of subtidal benthic invertebrate populations by restoring or recreating the natural marine habitats they live in. Habitat restoration or creation can come in various forms, for instance by restoring or creating the adequate substrate for specific species, by restoring or creating marine biogenic habitats (habitats created by the occurrence of specific marine species that form a new complex environment for other species to live in, such as coral reefs or kelp forests; Airoldi *et al*. 2008; Jones *et al.* 1994), or even by creating new artificial (man-made) structures aimed to locally enhance biodiversity (Clark & Edwards 1999). Here, descriptive studies of biodiversity on or around man-made structures already in place, such as oil rigs and wind farms, are not included, unless they were specifically deployed to enhance local diversity or left in place following decommissioning, to act as artificial reefs.

It should be kept in mind that habitat restoration and creation at a given site can be undertaken as a biodiversity offset strategy to replace the biodiversity lost at another impacted site, with the aim to achieve 'no net loss' of overall biodiversity (Ives & Bekessy 2015).

Airoldi L., Balata D. & Beck M.W. (2008) The gray zone: relationships between habitat loss and marine diversity and their applications in conservation. *Journal of Experimental Marine Biology and Ecology*, 366, 8–15.

Brooks T.M., Mittermeier R.A., Mittermeier C.G., Da Fonseca G.A., Rylands A.B., Konstant W.R., Flick P., Pilgrim J., Oldfield S., Magin G. & Hilton-Taylor C. (2002) Habitat loss and extinction in the hotspots of biodiversity. *Conservation Biology*, 16, 909–923.

Clark S. & Edwards A.J. (1999) An evaluation of artificial reef structures as tools for marine habitat rehabilitation in the Maldives. *Aquatic Conservation: Marine and Freshwater Ecosystems,* 9, 5–21.

Dobson A.P., Bradshaw A.D. & Baker A.Á. (1997) Hopes for the future: restoration ecology and conservation biology. *Science*, 277, 515–522.

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

Jones C.G., Lawton J.H. & Shachak M. (1994) Organisms as ecosystem engineers. Pages 130–147 in: *Ecosystem Management*. Springer, New York, NY.

Natural habitat restoration

12.1. Transplant captive-bred or hatchery-reared habitat-forming (biogenic) species

• We found no studies that evaluated the effects of transplanting captive-bred or hatchery-reared habitat-forming species on subtidal benthic invertebrate 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

Marine biogenic habitats are habitats created by the occurrence of a suite of specific marine species that form a new complex environment for other species to live in and can locally promote subtidal benthic invertebrate biodiversity. Such habitats include coral reefs, oyster reefs, mussel beds, and kelp forests (Jones *et al.* 1994). Restoring these habitats where they have been either degraded or lost can be achieved by transplanting new individuals of the biogenic species, for instance from captive-bred or hatchery-reared stock (McCay *et al*. 2003; Yap 2009). This technique can also be used to create new biogenic habitats where they do not naturally occur (Brumbaugh & Coen 2009). Transplanting biogenic species can promote subtidal benthic invertebrate biodiversity by providing additional habitat for species to colonize (Homziak *et al*. 1982).

Note that here, data on associate invertebrates are reported, but not on the transplanted species itself, which are reported in "Species management - Transplant captive-bred or hatchery-reared habitat-forming (biogenic) species". Related evidence from translocating studies of habitat-forming species are summarised under "Habitat restoration and creation – Translocate habitat-forming (biogenic) species". Other related evidence on biogenic habitat restoration is summarised under "Habitat restoration and creation – Restore biogenic habitats (other methods)".

Brumbaugh R.D. & Coen L.D. (2009) Contemporary approaches for small-scale oyster reef restoration to address substrate versus recruitment limitation: a review and comments relevant for the Olympia oyster, *Ostrea lurida* Carpenter 1864. *Journal of Shellfish Research*, 28, 147–161.

Homziak J., Fonseca M.S. & Kenworthy W.J. (1982) Macrobenthic community structure in a transplanted eelgrass (*Zostera marina*) meadow. *Marine Ecology Progress Series*, 211–221.

Jones C.G., Lawton J.H. & Shachak M. (1994) Organisms as ecosystem engineers. Pages 130–147 in: *Ecosystem Management*. Springer, New York, NY. McCay D.P.F., Peterson C.H., DeAlteris J.T. & Catena J. (2003) Restoration that targets function as opposed to structure: replacing lost bivalve production and filtration. *Marine Ecology Progress Series*, 264, 197–212.

Yap H.T. (2009) Local changes in community diversity after coral transplantation. *Marine Ecology Progress Series*, 374, 33–41.

12.2. Translocate habitat-forming (biogenic) species

Background

Marine biogenic habitats are habitats created by the occurrence of a suite of specific marine species that form a new complex environment for other species to live in, and which can locally promote subtidal benthic invertebrate biodiversity. Such habitats include coral reefs, oyster reefs, mussel beds, and kelp forests (Jones *et al.* 1994). Restoring these habitats where they have been either degraded or lost can be achieved by translocating new individuals of the biogenic species naturally occurring elsewhere, for instance from another healthy non-degraded site (Fariñas-Franco & Roberts 2014; Hughes *et al.* 2008). This technique can also be used to create new biogenic habitats where they do not naturally occur (Nelson *et al.* 2004).

Note that here, data on associated invertebrates are reported, but not on the translocated species itself, which are reported in "Species management – Translocate species". However, as the outcomes of translocating biogenic species can vary largely with the species and habitat that they form, studies have been grouped by habitat and/or wider taxonomic group (e.g: reefs or beds formed by molluscs such as oysters, mussels, snails; meadows made by seagrass; forests made by kelp; or reefs made by corals). Evidence from transplantation studies from hatchery-reared biogenic species are summarised under "Habitat restoration and creation – Transplant habitat-forming (biogenic) species" and under "Species management – Transplant/release captive-bred or hatchery-reared species".

Fariñas-Franco J.M. & Roberts D. (2014) Early faunal successional patterns in artificial reefs used for restoration of impacted biogenic habitats. *Hydrobiologia*, 727,75–94.

Hughes D.J., Poloczanska E.S. & Dodd J. (2008) Survivorship and tube growth of reef-building *Serpula vermicularis* (Polychaeta: Serpulidae) in two Scottish sea lochs. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 18, 117–129.

Jones C.G., Lawton J.H. & Shachak M. (1994) Organisms as ecosystem engineers. Pages 130–147 in: *Ecosystem Management*. Springer, New York, NY.

Nelson KA., Leonard L.A., Posey M.H., Alphin T.D. & Mallin M.A. (2004) Using transplanted oyster (*Crassostrea virginica*) beds to improve water quality in small tidal creeks: a pilot study. *Journal of Experimental Marine Biology and Ecology*, 298, 347–368.

12.2.1. Translocate reef- or bed-forming molluscs

- **Two studies** examined the effects of translocating habitat-forming molluscs on associated subtidal benthic invertebrate populations. Both were in Strangford Lough[1,2] (UK).

COMMUNITY RESPONSE (2 STUDIES)

- **Overall community composition (2 studies):** One replicated, site comparison study in Strangford Lough[1] found that plots with translocated mussels had different associated invertebrate communities to plots without mussels, but also to natural mussel beds. One replicated, controlled study in Strangford Lough[2] found that translocating mussels onto scallop shells or directly onto the seabed led to similar associated invertebrate communities.
- **Overall richness/diversity (2 studies):** One replicated, site comparison study in Strangford Lough[1] found that plots with translocated mussels had higher richness and diversity of associated invertebrates to plots without mussels, and similar to natural mussel beds. One replicated, controlled study in Strangford Lough[2] found that translocating mussels onto scallop shells or directly onto the seabed led to similar richness and diversity of associated invertebrates.

POPULATION RESPONSE (2 STUDIES)

- **Overall abundance (2 studies):** One replicated, site comparison study in Strangford Lough[1] presented unclear abundance results. One replicated, controlled study in Strangford Lough[2] found that translocating mussels onto scallop shells or directly onto the seabed led to higher abundance of associated invertebrates in one of two comparisons.

A replicated, site comparison study in 2010–2011 of 10 plots in Strangford Lough, Northern Ireland, UK (1 – same experimental set-up as 2) found that over a year after translocating habitat-forming horse mussel *Modiolus modiolus*, invertebrate species richness and diversity were higher in plots with translocated mussels than those without, and similar to those of nearby natural reefs. Species richness and diversity were reported as indices. All plots had different community composition from one another (community data presented as graphical analyses). The effect of translocation on invertebrate abundance was unclearly reported (see original paper). In 2010, divers translocated live adult horse mussels from nearby natural mussel patches within the Lough to four plots (1,000 mussels/plot). After 12 months, two quadrats (0.25 × 0.25 m) were deployed at each plot with translocated mussels and at four adjacent plots without translocated mussels. Sediment and shell were sampled in each quadrat to 10 cm depth. Organisms > 1 mm were identified and recorded as either counts or presence/absence. Natural horse mussel communities from two nearby horse mussel reefs within the lough were sampled in December 2010 using the same sampling methodology.

A replicated, controlled study in 2010–2011 of 12 plots in Strangford Lough, Northern Ireland, UK (2 – same experimental set-up as 1) found that over a year after translocating habitat-forming horse mussel *Modiolus modiolus*, overall invertebrate species richness and diversity increased, and invertebrate community composition changed, but with no differences between mussels translocated onto scallop shells or onto natural seabed. In plots where scallop shells had been added, either as elevated or flattened piles, and in plots where no shells were added, species richness and diversity (presented as indices) increased following translocation of horse mussels, but without differences between treatments. Community composition changed over time, but after a year was similar across treatments (data presented as graphical analyses). In addition, total abundance of invertebrates increased for the first six months but decreased between six and 12 months in all treatments. Over a year, abundance was higher in plots with elevated scallop shells (5–2,350 individuals) than in plots with flattened shells (2–1,370 individuals) or without shells (3–780 individuals). In November 2009–March 2010, sixteen tonnes of king scallop *Pecten maximus* shells were deployed in bags at four sites (17–19 m depth) to recreate suitable habitat for horse mussel reefs. Each site was divided into an elevated plot (8 m^2; shell rising 1 m above seabed) and a flattened plot (4 m^2; 0.5 m above seabed). Divers translocated live adult horse mussels from nearby natural mussel patches within the Lough into each plot and at four adjacent natural seabed plots without scallop shells (500 mussels/plot). One, six and 12 months after translocation, animals were identified and counted from one 0.5 × 0.5 m quadrat/plot. Strangford Lough is a marine protected area where fishing is prohibited.

(1) Fariñas-Franco J.M., Allcock L., Smyth D. & Roberts D. (2013) Community convergence and recruitment of keystone species as performance indicators of artificial reefs. *Journal of Sea Research,* 78, 59–74.
(2) Fariñas-Franco J.M. & Roberts D. (2014) Early faunal successional patterns in artificial reefs used for restoration of impacted biogenic habitats. *Hydrobiologia,* 727, 75–94.

12.2.2. Translocate reef-forming corals

• **Two studies** examined the effects of translocating habitat-forming corals on associated subtidal benthic invertebrate populations. One was in Tayabas Bay[1] (Philippines) and one in the South China Sea[2] (Philippines).

COMMUNITY RESPONSE (2 STUDIES)

- **Overall community composition (1 study):** One replicated, controlled, before-and-after study in the South China Sea[2] found that following coral translocation associated invertebrate communities did not change and remained similar to plots without translocated corals.
- **Overall richness/diversity (2 studies):** One replicated, controlled, before-and-after study in the South China Sea[2] found that following coral translocation richness of associated invertebrates increased but also increased in plots without corals, likely due to spill-over. One replicated, controlled study in Tayabas Bay[1] found that richness of associated invertebrates was higher in plots with translocated corals than in plots without.

POPULATION RESPONSE (1 STUDY)

- **Overall abundance (1 study):** One replicated, controlled, before-and-after study in the South China Sea[2] found that following coral translocation abundance of associated invertebrates increased and became higher than in plots without translocated corals.

A replicated, controlled study in 2000–2002 in five coral reef sites in Tayabas Bay, Philippines (1) found that plots with translocated corals developed higher invertebrate species richness than plots without corals, 9–27 months after translocation. After coral translocation, invertebrate species richness was higher in plots with corals (7–8 species) than in nearby and more distant plots without corals (3–6 species), but was lower than at the source site where the corals originated (10 species). Overall, 83-95% of translocated corals survived. Each of four sites of rocky seabed had eighteen 1 m^2 plots: six with translocated corals, six nearby without corals (interspersed with transplanted coral plots), and six 100 m away without corals. Between April 2000 and November 2001, three coral species were translocated from a nearby pristine reef (source site) to each translocated plot: *Acropora palifera* (2/plot), *Porites cylindrica* (2/plot), and *Porites lobata* (3/plot). In July 2002 (9–27 months after translocation), invertebrate species (excluding corals) were recorded during visual census by divers in all experimental plots, and in six plots at the source site.

A replicated, controlled, before-and-after study in 2010–2012 of nine plots in a restored coral reef off Santiago Island, northwestern Philippines, South China Sea (2) found that over the 19 months following translocation of corals, invertebrate species richness increased similarly at sites with and without translocated corals, abundance increased more at sites with than without corals, and community composition remained similar across all plots. Before translocation, all plots had similar species richness (0.3–0.5 species/plot), abundance (0.3–1.2/plot), and community composition (community data presented as graphical analyses). After 19 months, species richness had increased in all plots and was similar in plots with corals (3.0–3.3) and without (2.9). Abundance had increased in all plots but was higher in plots with corals (16–26) than without (3). Community composition remained similar in all plots after 19 months. After 19 months, 68–89% of translocated corals had survived. Increases in richness and abundance observed in plots without translocated corals were considered by authors to be due to spill-over effects from plots with translocated corals. Three clusters (50 m apart) of three plots (16 m^2; 5 m apart), were used for coral reef restoration. In each cluster, staghorn corals, *Acropora intermedia* and *Acropora pulchra,* were translocated to two plots (25 fragments/species in one, 50 fragments/species in the other), and one plot was left without corals. In July 2010 (before translocation), July 2011 (12 months after translocation), and February 2012 (19 months after translocation) divers visually identified and counted invertebrates belonging to six genera (see paper for details) in all plots.

(1) Yap H.T. (2009) Local changes in community diversity after coral transplantation. *Marine Ecology Progress Series*, 374, 33–41.
(2) dela Cruz D.W., Villanueva R.D. & Baria M.V.B. (2014) Community-based, low-tech method of restoring a lost thicket of *Acropora* corals. *ICES Journal of Marine Science*, 71, 1866–1875.

12.3. Restore biogenic habitats (other methods)

Background

Marine biogenic habitats are habitats created by the occurrence of a suite of specific marine species that form a new complex environment for other species to live in and can locally promote subtidal benthic invertebrate biodiversity. Such habitats include coral reefs, oyster reefs, mussel beds, and kelp forests (Jones *et al.* 1994). Restoring these habitats where they have been either degraded or lost can be achieved by transplanting or translocating new individuals of the biogenic species (evidence summarised under "Habitat restoration and creation – Transplant captive-bred or hatchery-reared habitat-forming (biogenic) species" and "Translocate habitat-forming (biogenic) species"; Fariñas-Franco & Roberts 2014; Yap 2009) or using other restoration methods that promote the natural recovery of the habitat. For instance, restoring seagrass beds by fertilizing the seabed and adding natural sediment to promote natural seagrass recovery can help the invertebrate community associated with seagrass beds to recover naturally over time (Bourque & Fourqurean 2014).

Bourque A.S. & Fourqurean J.W. (2014) Effects of common seagrass restoration methods on ecosystem structure in subtropical seagrass meadows. *Marine Environmental Research*, 97, 67–78.

Jones C.G., Lawton J.H. & Shachak M. (1994) Organisms as ecosystem engineers. Pages 130–147 in: *Ecosystem Management*. Springer, New York, NY. Fariñas-Franco J.M. & Roberts D. (2014) Early faunal successional patterns in artificial reefs used for restoration of impacted biogenic habitats. *Hydrobiologia*, 727,75–94.

Yap H.T. (2009) Local changes in community diversity after coral transplantation. *Marine Ecology Progress Series*, 374, 33–41.

12.3.1. Restore mussel beds

- **Two studies** examined the effects of restoring mussel beds (not by transplanting or translocating mussels) on mussels and mussel bed-associated subtidal benthic invertebrates. Both were in Strangford Lough[1a,b] (UK).

COMMUNITY RESPONSE (2 STUDIES)

- **Overall community composition (2 studies):** One replicated, controlled study in Strangford Lough[1a] found that after restoring beds of horse mussels by adding scallop shells to the seabed, overall invertebrate community composition in restored plots was different to that of unrestored plots. One replicated, controlled study in the same area[1b] found that after restoring beds of horse mussels by adding scallop shells to the seabed and translocating horse mussels, overall invertebrate community composition in plots restored with shells and mussels was different to plots restored without mussels (shells only), and both were different to unrestored plots and to nearby natural horse mussel beds.
- **Overall species richness/diversity (2 studies):** One replicated, controlled study in Strangford Lough[1a] found that after restoring beds of horse mussels by adding scallop shells to the seabed, overall invertebrate species diversity was lower in restored plots compared to unrestored plots, but species richness was similar. One replicated, controlled study in the same area[1b] found that after restoring beds of horse mussels by adding scallop shells to the seabed and translocating horse mussels, species richness and diversity were higher in restored plots with mussels and shells compared to plots with shells only, and similar to nearby natural horse mussel beds.

POPULATION RESPONSE (1 STUDY)

- **Overall abundance (1 study):** One replicated, controlled study in Strangford Lough[1a] found that after restoring beds of horse mussels by adding scallop shells to the seabed, overall invertebrate abundance was higher in restored plots compared to unrestored plots.

A replicated, controlled study in 2010–2011 of 12 plots in Strangford Lough, Northern Ireland, UK (1a – same experimental set-up as 1b), found that one year after restoring horse mussel *Modiolus modiolus* habitat by adding scallop shells to the seabed, invertebrate community composition in restored plots was different to that of unrestored plots. Community data were reported as graphical analyses. In addition, while total invertebrate abundance was higher in restored plots (258–830 individuals) compared to unrestored plots (40–58 individuals), species diversity was lower in restored plots (data reported as diversity indices). Species richness was similar across plots (data reported as indices). Within restored plots, there were no differences between plots with elevated scallop shells and plots with flattened shells. In 2010 sixteen tonnes of scallop shells were deployed in bags at four sites (17–19 m depth) to recreate suitable habitat for horse mussel reefs. Each site was divided into an elevated plot (8 m^2; shell rising 1 m above seabed) and a flattened plot (4 m^2; 0.5 m above seabed). After 12 months, one quadrat (0.25 × 0.25 m) was deployed at each plot and at four adjacent unrestored plots. Sediment and shell were sampled for each quadrat to 10 cm depth. Organisms (>1 mm) were identified and recorded as either counts or presence/absence.

A replicated, controlled study in 2010–2011 of multiple plots in Strangford Lough, Northern Ireland, UK (1b – same experimental set-up as 1a), found that one year after restoring horse mussel *Modiolus modiolus* biogenic habitat by adding scallop shells to the seabed and translocating horse mussels, overall invertebrate community composition in plots restored with shells and mussels was different to plots restored without mussels (shells only) and both were different to unrestored plots. Community data were presented as graphical analyses. In addition, species richness and diversity were higher in restored plots with mussels and shells compared to plots with shells only (data reported as indices). When compared with nearby natural horse mussel reefs, restored plots (with shells and mussels) had different community composition, despite having similar species richness and diversity. Within restored plots after a year, there were no differences between plots with elevated scallop shells and plots with flattened shells, apart for translocated mussel mortality which was lower in elevated plots (5%) compared to flattened plots or unrestored plots (19%). In 2010 sixteen tonnes of scallop shells were deployed in bags at four sites (17–19 m depth) to recreate suitable habitat for horse mussel reefs. Each site was divided into an elevated plot (8 m^2; shell rising 1 m above seabed) and a flattened plot (4 m^2; 0.5 m above seabed). Each plot was then sub-divided, and divers translocated live adult horse mussels from nearby natural mussel patches within the Lough into on half of each plot (500 mussels/subplot). After 12 months, one quadrat (0.25 × 0.25 m) was deployed at each subplot and at four adjacent unrestored plots. Sediment and shell were sampled for each quadrat to 10 cm depth. Organisms (>1 mm) were identified and recorded as either counts or presence/absence. Natural horse mussel communities from two nearby horse mussel reefs within the Lough were sampled in December 2010 using the same sampling methodology.

(1a,b) Fariñas-Franco J.M., Allcock L., Smyth D. & Roberts D. (2013) Community convergence and recruitment of keystone species as performance indicators of artificial reefs. *Journal of Sea Research,* 78, 59–74.

12.3.2. Restore oyster reefs

- **Eight studies** examined the effects of restoring oyster reefs (not by transplanting or translocating oysters) on oysters and oyster reef-associated subtidal benthic invertebrates. Two were in the Gulf of

Mexico[1,2] (USA), one was a global review[3], four were in the North Pacific Ocean[4a-d] (USA), and one was in the Mission-Aransas estuary[5] (USA).

COMMUNITY RESPONSE (3 STUDIES)

- **Overall community composition (2 studies):** One of two replicated, controlled studies in the Gulf of Mexico[2] and the Mission-Aransas estuary[5] found that after restoring eastern oyster reefs, the community composition of combined mobile decapod invertebrates and fish was similar on all types of restoration material used[2], but the other found that composition varied with the material used[5].
- **Overall species richness/diversity (3 studies):** One replicated, site comparison study in the Gulf of Mexico[1] found that diversity of reef-associated invertebrates was similar in reefs restored by laying rocks regardless of age, in young reefs restored by laying oyster shells, and in natural reefs, but lower in old shell-restored reefs. One replicated, controlled study in the Gulf of Mexico[2] found that diversity of reef-associated invertebrates was higher in all restored reefs than on unrestored sediment, but that diversity varied between the restoration materials used. One replicated, controlled study in the Mission-Aransas estuary[5] found that diversity of fish, crabs and shrimps varied with the restoration material used.

POPULATION RESPONSE (7 STUDIES)

- **Overall abundance (2 studies):** One replicated, site comparison study in the Gulf of Mexico[1] found that the effect of restoring eastern oyster reefs on the abundance of reef-associated invertebrates depended on the material used for restoration and the age of the reef. One replicated, controlled study in the Gulf of Mexico[2] found that abundance of combined reef-associated mobile decapod invertebrate and fish was similar on all restored reefs regardless of the restoration material used, and higher than on unrestored sediment.
- **Crustacean abundance (1 study):** One replicated, controlled study in the Mission-Aransas estuary[5] found that after restoring eastern oyster reefs, crab abundance, but not biomass, and shrimp biomass, but not abundance, varied with the restoration material used.
- **Oyster abundance (6 studies):** One replicated, site comparison study in the Gulf of Mexico[1] found that oyster reefs restored by laying rocks had similar oyster abundance to natural reefs, and higher than reefs restored by laying oyster shells. One replicated, controlled study in the Mission-Aransas estuary[5] found that oyster cover and abundance varied with the restoration material used. One replicated, controlled study in the Gulf of Mexico[2] found that oyster spat abundance was similar on all types of restoration material used, and higher than on unrestored sediment. Three replicated, controlled studies in the North Pacific Ocean[4a-c] found that restoring oyster reefs by placing lines of clam shells below Mean Lower Low Water (MLLW) led to higher cover of clam shells by oysters than when placing the lines above MLLW[4a], that for those placed below MLLW, keeping them there led to similar cover compared to moving them above MLLW halfway through the study[4b], and that placing the lines on cobbly seabed led to similar cover compared to placing them on muddy seabed[4c].
- **Oyster reproductive success (3 studies):** Three replicated, controlled studies in the North Pacific Ocean[4a,c,d] found that restoring oyster reefs by placing lines of clam shells below Mean Lower Low Water (MLLW) led to higher recruitment of oyster spat on clam shells than by lacing lines above MLLW[4a], that recruitment was higher on lines placed on cobbly seabed than on muddy seabed[4c], and that recruitment was similar on lines placed near or far from the nearest adult oyster populations[4d].
- **Oyster survival (5 studies):** One global systematic review[3] found that two of nine restoration techniques (restoring oyster reef by transplanting juveniles, and by creating no-harvest sanctuaries) assessed resulted in over 85% survival of restored oysters. Four replicated, controlled studies in the North Pacific Ocean[4a-d] found that restoring oyster reefs by placing lines of clam shells below Mean Lower Low Water (MLLW) led to similar survival of oysters than when

placing the lines above MLLW[4a], but that for those placed below MLLW, moving them above MLLW halfway through the study led to higher survival than keeping then below[4b], that survival was similar on lines placed on cobbly seabed or muddy seabed[4c], and that survival was similar on lines placed near or far from the nearest adult oyster populations[4d].

- **Oyster condition (5 studies):** One replicated, controlled study in the Gulf of Mexico[2] found that the effect of restoring eastern oyster reefs on average spat size varied with the restoration material used. One replicated, controlled study in the North Pacific Ocean[4a] found that restoring oyster reefs by placing lines of clam shells below Mean Lower Low Water (MLLW) led to similar growth of oysters on the shells than placing lines above MLLW. Four replicated, controlled studies in the North Pacific Ocean[4a-d] found that restoring oyster reefs by placing lines of clam shells below Mean Lower Low Water (MLLW) led to higher cover of clam shells by non-native species than placing lines above MLLW[4a], but that for those placed below MLLW, moving them above MLLW halfway through the study led to lower cover than keeping then below[4b], that cover was similar on lines placed on cobbly seabed or muddy seabed[4c], and that cover of clam shells by non-native species was higher on lines placed near compared to far from the nearest adult oyster populations[4d].

A replicated, site comparison study in 2011 of 20 oyster reefs in the northern Gulf of Mexico, from Texas to Florida, USA (1) found that the effect of restoring reefs of eastern oyster *Crassostrea virginica* on oysters and reef-associated invertebrates depended on the material used for restoration and the age of the reef. Reefs restored by laying rocks had similar oyster abundance (102–105 oyster/m^2) to natural reefs (136 oyster/m^2), while reefs restored by laying oyster shells had lower oyster abundance (3–22 oyster/m^2) than any other reefs, regardless of the age of the restored reefs. In addition, diversity of reef-associated invertebrates (reported as diversity index) was similar in rock-restored reefs regardless of age, young shell-restored reefs (under five-year-old) and natural reefs, but significantly lower in old shell-restored reefs (over five-year-old). Overall abundance of reef-associated invertebrates was similar in young rock-restored reefs (106) and old shell-restored reefs (58) to natural reefs (182), but higher in old rock-restored reefs (345) and young shell-restored reefs (338) compared to natural reefs. Invertebrates were surveyed on 20 reefs (100 m offshore; approximately 2 m depth). Eight had been restored by laying rocks (six old; two young), five had been restored by laying shells (two old; three young), and seven were natural reefs. In October 2011, divers counted live eastern oysters in the top 10 cm of reef (five 0.25 m^2 quadrats/reef). In May and again in July 2011, two 30 ×30 cm bags containing oyster shells were deployed at each reef to capture invertebrates and retrieved after one month (80 bags total). All four bags/reef were combined, and invertebrates (<1 mm) were identified and counted.

A replicated, controlled study (year unspecified) of 10 soft seabed sites in the Gulf of Mexico, Texas, USA (2) found that the effect of restoring reefs of eastern oyster *Crassostrea virginica* on oysters and reef-associated mobile decapod invertebrates and fish depended on the material used for restoration. Average oyster spat abundance was similar on all types of restoration material used (840–1,390 spat/m^2) and higher than at unrestored sediment (0 spat/m^2). Average spat size was higher on concrete, river rock and oyster shell (15.5–15.8 mm) than on limestone (13.2 mm) and porcelain (11.8 mm). The community composition of combined mobile decapod invertebrates and fish was similar on all types of restoration material used (community data reported as graphical analyses). Average abundance of mobile decapod invertebrates and fish was similar on all types of restoration material used (310–550 individual/m^2) and higher than on unrestored sediment (4.5 individual/m^2). Diversity was higher on any restoration material than on unrestored sediment (data reported as a diversity index). Between restoration materials, diversity was higher on porcelain and oyster shell than concrete, which were all higher than on river rock and limestone. Five trays (0.75 m^2) were deployed at each of 10 sites and filled with one of five types of restoration material (concrete, porcelain, limestone, river rock, oyster shell). After four months, all trays and one bare (unrestored) sediment patch were collected using a 1 m^2 grab with a 1.6 mm mesh. One 0.09 m^2 quadrat was deployed on each tray. Eastern oyster spat were counted in grab and quadrat samples. The shell height of up to 20 spat/tray was measured using callipers. Mobile decapod invertebrates and fish were identified and counted.

A systematic review conducted in 2014 of studies from across the world (3) found that following oyster reef restoration projects, the survival of oysters varied with the restoration technique used. Comparing nine different techniques, the survival of oysters varied between 0% and 100% (survival for each technique not shown). Two of the nine restoration techniques (restoring oyster reef by transplanting juveniles, and by creating no-harvest sanctuaries) resulted in over 85% survival of restored oysters. A systematic review of the literature available by 21 November 2014 on the feasibility, survival, and costs of oyster reef restoration was conducted. Out of the 81 studies found on oyster reef restoration, 24 studies were included in the systematic review. Data on the restoration technique used and survival of "restored oysters" (exact definition not stated) were extracted and analysed.

A replicated, controlled study in 2012–2014 of seven estuarine sites in Monterey Bay, North Pacific Ocean, USA (4a) found that the effects on oysters and non-native species of restoring the reef-forming Olympia oyster *Ostrea lurida* by placing lines of clam shells (for oysters to settle on) varied with tidal elevation. After five months, oyster recruitment was higher on lines placed below Mean Lower Low Water (MLLW) (122 oysters/line) compared to lines placed above MLLW (70 oysters/line). Survival after five months was similar at both tidal elevation (below: 89%; above: 86%), leading to higher cover of clam shells by oysters below MLLW (20%) than above (12%). However, cover by non-native species was higher below MLLW (25%) than above (20%). In addition, after two years, oysters reached similar average maximum size below (60 mm) and above (59 mm) MLLW. In July 2012, six lines of clam shells were deployed at each of seven sites (at least 1 km apart): three lines above MLLW, three below. In December 2012, live and dead oysters were counted on each line, and the percentage cover of clam shells by oysters and non-native species (sponges, tunicates, bryozoans, and hydrozoans) was visually assessed. In May 2014, the five largest oysters on each line were measured.

A replicated, controlled study in 2012–2014 of four estuarine sites in Monterey Bay, North Pacific Ocean, USA (4b) found that the effects) on oysters and non-native species of restoring the reef-forming Olympia oyster *Ostrea lurida* by placing lines of clam shells (for oysters to settle on depended on if the lines were moved above or remained below Mean Lower Low Water (MLLW). After a year, cover of clam shells by oysters was similar on lines moved above (41%) and lines which had remained below (44%) MLLW. However, oyster survival after a year was higher on lines moved above (94%) than lines remained below (77%) MLLW, and clam shell cover by non-native species was lower (above: 4%; below: 43%). In July 2012, two lines of clam shells were deployed at each of four sites (at least 1 km apart) below MLLW. In June 2013, at each site, one line was moved above MLLW. In May 2014, live and dead oysters were counted on each line, and the percentage cover of clam shells by oysters and non-native species (sponges, tunicates, bryozoans, and hydrozoans) was visually assessed. No oyster recruitment occurred in 2013.

A replicated, controlled study in 2012–2014 of four estuarine sites in Monterey Bay, North Pacific Ocean, USA (4c) found that the effects on oysters and non-native species of restoring the reef-forming Olympia oyster *Ostrea lurida* by placing lines of clam shells (for oysters to settle on) varied with seabed type. After five months, oyster recruitment was higher on lines placed on cobbly seabed (138 oysters/line) compared to lines placed on muddy seabed (83 oysters/line). Survival after five months was lower on cobbly seabed (61%) than muddy seabed (99%). This led to similar cover of clam shells by oysters on cobbly (15%) and muddy seabed (18%). In addition, cover by non-native species was similar on both seabed type (cobble: 26%; mud: 31%). In July 2012, six lines of clam shells were deployed at each of four sites (at least 1 km apart): two cobbly sites and two muddy sites. In December 2012, live and dead oysters were counted on each line, and the percentage cover of clam shells by oysters and non-native species (sponges, tunicates, bryozoans, and hydrozoans) was visually assessed.

A replicated, controlled study in 2012–2014 of five muddy estuarine sites in Monterey Bay, North Pacific Ocean, USA (4d) found that the effects on oysters and non-native species of restoring the reef-forming Olympia oyster *Ostrea lurida* by placing lines of clam shells (for oysters to settle on) varied with distance to the nearest adult oyster populations. After five months, oyster recruitment and survival were similar on lines placed near (adjacent; 83 oysters/line; 14% survival) and far (over 300 m away; 77 oysters/line; 15%) from the nearest adult oyster populations. However, cover by non-native species was higher at sites near (31%) than far (14%) from the nearer adult populations. In July 2012, six lines of clam shells were deployed at each of five sites (at least 1 km apart): two sites "near" and three "far" from the nearest adult populations. In December 2012, live and dead oysters were counted on each line, and the percentage cover of clam shells by oysters and non-native species (sponges, tunicates, bryozoans, and hydrozoans) was visually assessed.

A replicated, controlled study in 2013–2015 of 12 restored reefs in the Mission-Aransas estuary, southern coast of Texas, USA (5) found that the effects of restoring reefs of eastern oyster *Crassostrea virginica* on oysters and reef-associated organisms, after 21 months, depended on the material used. After 21 months, the community structure of combined invertebrates and fish differed with material (data presented as a graphical analysis). The diversity of mobile organisms (fish, crabs and shrimps) was similar across material (reported as a diversity index). Oysters dominated the cover of sessile organisms on all reefs, but cover was lower on river rock (41%) compared to all other material (68–53%). Oyster abundance was higher on concrete (1,020/m^2), than limestone (940/m^2), oyster shell (830/m^2), and river rock (600/m^2). Crabs (five species combined) dominated the mobile organisms across reefs, with no effect of material on their abundance (270–440/m^2). Crab biomass was higher on oyster shell (53 g/m^2) and concrete (38 g/m^2) than river rock (24 g/m^2), but not limestone (36 g/m^2). Shrimps (five species combined) were more abundant on oyster shell (140/m^2) than any other material (60–120/m^2). Shrimp biomass was similar on all material (3–6 g/m^2). In 2013, twelve oyster reefs (152 m^3) were constructed with either concrete, river rocks, limestones, or oyster shells (3 reefs/material). After three months, six trays filled with 19 L of matching material were deployed at each reef. Quarterly, one tray/reef was retrieved and mobile organisms (> 4mm) identified, counted, and dry-weighed. Oysters were counted, and their percentage cover assessed. In addition, other sessile invertebrates were assessed, and a benefit-cost ratio for each material was calculated (see paper).

(1) Brown L.A., Furlong J.N., Brown K.M. & La Peyre M.K. (2014) Oyster reef restoration in the northern Gulf of Mexico: effect of artificial substrate and age on nekton and benthic macroinvertebrate assemblage use. *Restoration Ecology*, 22, 214–222.

(2) George L.M., De Santiago K., Palmer T.A. & Pollack J.B. (2015) Oyster reef restoration: effect of alternative substrates on oyster recruitment and nekton habitat use. *Journal of Coastal Conservation*, 19, 13–22.

(3) Bayraktarov E., Saunders M.I., Abdullah S., Mills M., Beher J., Possingham H.P., Mumby P.J. & Lovelock C.E. (2016) The cost and feasibility of marine coastal restoration. *Ecological Applications*, 26, 1055–1074.

(4a-d) Zabin C.J., Wasson K. & Fork S. (2016) Restoration of native oysters in a highly invaded estuary. *Biological Conservation*, 202, 78–87.

(5) Graham P.M., Palmer T.A. & Beseres Pollack J. (2017) Oyster reef restoration: substrate suitability may depend on specific restoration goals. *Restoration Ecology*, 25, 459–470.

12.3.3. Restore seagrass beds/meadows

- **Three studies** examined the effects of restoring seagrass beds (not by transplanting or translocating seagrass) on seagrass bed-associated subtidal benthic invertebrates. One was in the North Atlantic Ocean[1] (USA), one in the Indian Ocean[2] (Kenya), and one in the Florida Keys[3] (USA).

COMMUNITY RESPONSE (2 STUDIES)

- **Overall community composition (1 study):** One replicated, randomized, controlled study in the Florida Keys[3] found that restoring seagrass beds by fertilizing the seabed had no effect on overall invertebrate community composition, but adding sand led to communities different from both unrestored and natural sites.
- **Overall species richness/diversity (2 studies):** One replicated, randomized, controlled study in the Florida Keys[3] found that after restoring seagrass beds by fertilizing the seabed and adding sand, overall invertebrate species richness was similar at restored, unrestored, and natural sites. One replicated, controlled study in the Indian Ocean[2] found that transplanting plastic seagrass mimics into bare sites, previously-restored seagrass sites, and natural seagrass sites, resulted in similar invertebrate diversity on mimic leaves and in the surrounding sediment, and similar species richness on mimic leaves at all restored sites as on natural seagrass leaves.

POPULATION RESPONSE (3 STUDIES)

- **Overall abundance (3 studies):** One replicated, randomized, controlled, before-and-after study in the North Atlantic Ocean[1] found that after restoring seagrass beds, the abundance of mobile invertebrates had increased and was higher in restored than unrestored plots, but the abundance of sessile invertebrates had not increased. One replicated, controlled study in the Indian Ocean[2] found that transplanting plastic seagrass mimics into bare sites, previously-restored seagrass sites, and natural seagrass sites, resulted in similar abundance of invertebrate in the surrounding sediment across sites, and resulted in different abundance of invertebrates on mimic leaves between sites although all had lower abundances than on natural seagrass leaves. One randomized, replicated, controlled study in the Florida Keys[3] found that after restoring seagrass beds by fertilizing the seabed or adding sand, overall invertebrate abundance was not different at restored sites compared to both unrestored and natural sites.

A replicated, randomized, controlled, before-and-after study in 1990 of eight estuarine plots in Waquoit Bay, Massachusetts, North Atlantic Ocean, USA (1) found that four months after restoration of common eelgrass *Zostera marina* beds by removing macroalgae, abundance of mobile invertebrates (decapods) had increased and was higher in restored than unrestored plots, but abundance of sessile invertebrates had not increased. Prior to restoration, all plots had similar abundance of decapods (restored: 24 vs unrestored: 27/m^2) and sessile invertebrates (26,000 vs 22,000/m^2). After four months, decapod abundance had increased and was higher in restored (82) than unrestored plots (35/m^2), while sessile invertebrate abundance had decreased overall and remained similar across plots (restored: 5,000; unrestored: 8,000/m^2). Following restoration, macroalgae biomass decreased in all plots (restored: from 96 to 20; unrestored: from 127 to 33 g/m^2), and eelgrass abundance increased in all plots and was higher in restored plots (restored: from 3 to 19; unrestored: from 3 to 7 shoots/m^2). Two 25 × 25 m experimental areas were selected, each with four 100 m^2 plots. In May 1990, eelgrass was restored in one randomly selected area by manually removing macroalgae weekly. A second area was left unrestored. Plots were surveyed in April and monthly in June–September. Sessile invertebrates (>0.5 mm) were sampled in 3 plots/area using two cores (0.073 m^2) and counted. Macroalgae were obtained from the cores and dry-weighed. Decapods (> 3 mm) were sampled using one 1 m^2 throw net/plot and counted. Eelgrass shoots were counted along one 14 m^2 diagonal transect/plot.

A replicated, controlled study in 2008 of 12 seagrass sites in Diani Beach, Indian Ocean, south coast of Kenya (2) found that transplanting plastic seagrass mimics into either bare sites, previously-restored seagrass sites, or natural seagrass sites, resulted in different abundance of invertebrates on mimic leaves between sites after 21 days, but similar diversity of invertebrates on mimic leaves, and similar diversity and abundance of invertebrates in the surrounding sediment (values not reported). Abundance on mimic leaves was higher on the natural (39 individuals/100 cm^2) and bare (49) sites, compared to the previously-restored sites (12), but remained lower on all mimic leaves compared to natural seagrass leaves (83). However, species richness on all mimic leaves (10) appeared similar to that of natural seagrass leaves (11) (not statistically tested). Invertebrate abundances in the sediment were not reported. There were nine sites (0.7 m depth): three bare (no natural seagrass), three previously-restored (one year previously by transplanting natural seagrass, but damaged overwinter), and three natural seagrass sites. Three plastic seagrass mimics (cluster of four plants) were transplanted to each site in August 2008. After 21 days, invertebrates (38 μm–1 mm in size) living on the leaves and in the sediment around each mimic were identified and counted (see paper for details). Three natural seagrass sites without transplanted mimics were sampled for comparison.

A replicated, randomized, controlled study in 2010–2011 of 18 seagrass restoration sites in Cutter Bank, Florida Keys, USA (3) found that one year after restoration, *Thalassia testudinum* seagrass beds developed a different invertebrate community composition compared to unrestored sites, but not similar to that of natural sites, and similar species richness and abundance to unrestored and natural sites. Restoring by fertilizing had no effect on the invertebrate community but adding sand led to communities different from both unrestored and natural sites (community data presented as graphical analyses and statistical model results). After a year, species richness was similar across sites (restored: 8.6; unrestored: 7.5; natural: 10.2 species). Invertebrate abundance was not different at sites restored by adding sand (abundance: 43.6 individuals/sample) compared to both unrestored (35.7) and natural sites (38.6), with no effect of fertilizing (data not shown).

Abundance at the natural sites had declined throughout the year (from 92.2 to 38.6). Seagrass cover declined at all sites throughout the year. Seagrass restoration was undertaken in 2010 at 12 sites disturbed by vessels and six undisturbed natural sites. Three disturbed sites were left unrestored, three were restored by fertilizing, three by adding sand, and three by both fertilizing and adding sand. Of the six undisturbed natural sites, three were fertilized, and three were left natural. Restoration option was randomly allocated to disturbed and undisturbed sites. Zero, three, six and 12 months after restoration, three sediment samples were collected at each site using a randomly placed hand corer (7.3 × 10 cm). Invertebrates (>0.5 mm) were identified and counted.

(1) Deegan L.A., Wright A., Ayvazian S.G., Finn J.T., Golden H., Merson R.R. & Harrison J. (2002) Nitrogen loading alters seagrass ecosystem structure and support of higher trophic levels. A*quatic Conservation: Marine and Freshwater Ecosystems*, 12, 193–212.
(2) Daudi L.N., Uku J.N. & De Troch M. (2013) Role of the source community for the recovery of seagrass associated meiofauna: a field colonisation experiment with seagrass mimics in Diani Beach, Kenya. *African Journal of Marine Science*, 35, 1–8.
(3) Bourque A.S. & Fourqurean J.W. (2014) Effects of common seagrass restoration methods on ecosystem structure in subtropical seagrass meadows. *Marine Environmental Research*, 97, 67–78.

12.4. Restore coastal lagoons

- **Three studies** examined the effects restoring coastal lagoons on subtidal benthic invertebrate populations. One study was in the Chilika lagoon[1] (India), and two in East Harbor lagoon[2,3] (USA).

COMMUNITY RESPONSE (3 STUDIES)

- **Crustacean richness/diversity (1 study):** One before-and-after study in Chilika lagoon[1] found that following hydrological restoration total crustacean species richness decreased, but changes varied with species groups (decreases in prawn and crab species; increases in lobster species). The lagoon also hosted new species not found before.
- **Mollusc richness/diversity (2 studies):** Two studies in East Harbor lagoon[2,3] found that following hydrological restoration molluscs recolonised the lagoon and their species richness increased in the first three years[2] but decreased over the following six[3].

POPULATION RESPONSE (3 STUDIES)

- **Crustacean abundance (1 study):** One before-and-after study in Chilika lagoon[1] found that following hydrological restoration abundances of prawns and crabs increased.
- **Mollusc abundance (2 studies):** Two studies in East Harbor lagoon[2,3] found that following hydrological restoration molluscs recolonised the lagoon and their total abundance increased in the first three years[2], but later decreased over the following six years[3].

Background

Coastal lagoons (areas of shallow, coastal salt water, wholly or partially separated from the sea by sandbanks, shingle or, less frequently, rocks) are highly biodiverse systems. They can also be connected to land water through rivers, and as such host both marine, brackish, and freshwater invertebrate species. However, coastal lagoons are highly threatened by eutrophication, pollution, urbanization, and diverse forms of modification in their watersheds, caused by high levels of human activity in the coastal zones of all continents (Esteves *et al*. 2008), resulting in biodiversity loss. Coastal lagoons could be restored through various means, including re-salination, hydrological modifications and algae harvesting (Ghosh *et al*. 2006; Lenzi *et al*. 2003). Restoring coastal lagoons can potentially help promote local biodiversity and recover lost or declining species of subtidal benthic invertebrates (Mohapatra *et al.* 2007).

Esteves F.A., Caliman A., Santangelo J.M., Guariento R.D., Farjalla V.F. & Bozelli R.L. (2008) Neotropical coastal lagoons: an appraisal of their biodiversity, functioning, threats and conservation management. *Brazilian Journal of Biology*, 68, 967–981.

Ghosh A.K., Pattnaik A.K. & Ballatore T.J. (2006) Chilika Lagoon: Restoring ecological balance and livelihoods through re-salinization. *Lakes & Reservoirs: Research & Management*, 11, 239–255.

Lenzi M., Palmieri R. & Porrello S. (2003) Restoration of the eutrophic Orbetello lagoon (Tyrrhenian Sea, Italy): water quality management. *Marine Pollution Bulletin*, 46, 1540–1548.

Mohapatra A., Mohanty R.K., Mohanty S.K., Bhatta K.S. & Das N.R. (2007) Fisheries enhancement and biodiversity assessment of fish, prawn and mud crab in Chilika lagoon through hydrological intervention. *Wetlands Ecology and Management*, 15, 229–251.

A before-and-after study in 1996–2004 in a degraded lagoon connected to the Bay of Bengal, east coast of India (1) found that, four years after restoring its hydrology, crustacean species richness decreased, but abundance of commercially valued crustaceans increased. There were reductions in the number of prawn species (before: 24; after: 18 but four were new) and crab species (before: 28; after: 14 but seven were new), and an increase in lobster species (before: 0; after: 2). Abundance (as four-year averages of commercial landings) increased by 1,200% for prawns (before: 187; after: 2,430 t), and 1,135% for crabs (before: 10; after: 130 t). No commercial landings were reported for lobsters. Authors report that increases in landings were correlated with increases in salinity after restoration. The ecological status of the Chilika lagoon declined throughout the 20th century. In 2000, channels were dredged or extended to increase connections to the sea and rivers and improve the hydrology. Data were obtained from Orissa state Department of Fisheries for 1996–2000 (pre-restoration), and by the authors for 2000–2004 (post-restoration), following the same sampling methods. Thirty-four landings centres were visited monthly and prawn and crab catches, including the commercially valued species *Penaeus monodon, Penaeus indicus, Metapenaeus monoceros, Metapenaeus dobsoni, Macrobrachium sp.* and *Scylla sp.,* were recorded (see study for details).

A study in 2005 in a lagoon connected to Cape Cod Bay, Massachusetts, USA (2 – same experimental set-up as 3) found that, three years after restoring its connection to the sea, molluscs had recolonised the lagoon, but molluscan abundance and species richness significantly varied within the lagoon, increasing with salinity and proximity to its connection to the sea. Sixteen molluscan species were recorded across the lagoon. Species richness in Moon Pond (13 species; highest salinity; closest to the sea) was significantly higher than in the central lagoon (9 species; intermediate salinity and distance to the sea) and the northwest cove (2 species, lowest salinity and furthest from the sea). Total mollusc abundance varied spatially within the lagoon, from 0.3 to 3,470 individuals/m^2. Abundance of four selected species (softshell clam *Mya arenaria*; northern quahog *Mercenaria mercenaria*; blue mussel *Mytilus edulis;* periwinkle *Littorina sp.*) followed the same spatial pattern as species richness (see paper for details). In 2002, tidal flow was partially restored to East Harbor lagoon (dominated by freshwater) by opening a culvert connecting to Cape Cod. Previously, no molluscan species were reported. In summer 2005, locations within three areas of the lagoon were surveyed twice (Moon Pond: 10 locations; central lagoon: 30 locations; northwest cove: 10 locations) using cores (0.79 m^2) and quadrats (0.45 m^2). Molluscs (>2 mm in cores; >0.64 cm in quadrats) were identified and counted. Salinity was measured at each location.

A study in 2005–2011 in a lagoon connected to Cape Cod Bay, Massachusetts, USA (3 – same experimental set-up as 2) found that, between three and nine years after restoring its connection to the sea, species richness and abundances of molluscs that had recolonised the lagoon following reconnection were decreasing over time, but effects varied geographically within the lagoon. Species richness decreased from 16 in 2005 to eight in 2011 across the lagoon, due to significant decreases in Moon Pond (from 14 to 5). Abundance of mollusc species also declined over time (total values not provided, see paper for details on each species abundance). Abundance of the softshell clam *Mya arenaria* (the dominant species in the lagoon and only one present in all areas each year), declined between 2005 and 2011, from 3,200/m^2 to 8/m^2 in Moon Pond and from 2,900/m^2 to 7/m^2 in the central lagoon, and remained low in northwest cove (0.2/m^2 in 2005, 1/m^2 in 2011). In 2002, tidal flow was partially restored to East Harbor lagoon (dominated by freshwater) by opening a culvert connecting to Cape Cod. Previously, no molluscan species were reported. In summer 2007, 2008 and 2011, locations within three areas of the lagoon were surveyed (Moon Pond: 20 locations; central lagoon: 24–30 locations; northwest cove: 4–15 locations) using cores (0.79 m^2). Molluscs (>1 mm) were identified and counted. Data were compared to 2005 data from a previous study by Thelen & Thiel (2009) (summarised in (2)).

(1) Mohapatra A., Mohanty R.K., Mohanty S.K., Bhatta K.S. & Das N.R. (2007) Fisheries enhancement and biodiversity assessment of fish, prawn and mud crab in Chilika lagoon through hydrological intervention. *Wetlands Ecology and Management*, 15, 229–251.
(2) Thelen B.A. & Thiet R.K. (2009) Molluscan community recovery following partial tidal restoration of a New England estuary. *Restoration Ecology,* 17, 695–703.
(3) Thiet R.K., Kidd E., Wennemer J.M. & Smith S.M. (2014) Molluscan community recovery in a New England back-barrier salt marsh lagoon 10 years after partial restoration. *Restoration Ecology*, 22, 447–455.

12.5. Refill disused borrow pits

- **One study** examined the effects of refilling disused borrow pits on subtidal benthic invertebrate populations. The study was in Barnegat Bay estuary[1] (USA).

COMMUNITY RESPONSE (1 STUDY)

- **Overall richness/diversity (1 study):** One before-and-after, site comparison study in Barnegat Bay estuary[1] found that overall invertebrate species richness and diversity increased at a disused borrow pit after being refilled with sediments but remained lower than at a natural non-dredged site.

POPULATION RESPONSE (1 STUDY)

- **Overall abundance (1 study):** One before-and-after, site comparison study in Barnegat Bay estuary[1] found that overall invertebrate abundance increased at a disused borrow pit after being refilled with sediments but remained lower than at a natural non-dredged site.

Background

'Aggregates' is the collective term for sand, gravel and crushed rock. They are used as raw materials for the construction industry as well as for beach replenishment schemes (De Groot 1996). Extraction leads to physical impacts and disruption of the seabed, including the formation of borrow pits, also known as dredged holes (Reine *et al*. 2013). Borrow pits tend to be very different from the surrounding natural seabed, being much deeper, isolated, and of different sediment type. The natural seabed ecology could be restored by refilling the disused borrow pits following cessation of extraction (Reine *et al.* 2013). This can potentially help the recolonization and natural recovery of the invertebrate community.

Evidence for interventions related to aggregate extraction is summarised under "Threat: Energy production and mining – Mining and quarrying".

De Groot S.J. (1996) The physical impact of marine aggregate extraction in the North Sea. *ICES Journal of Marine Science*, 53, 1051–1053.

Reine K., Clarke D., Ray G. & Dickerson C. (2013) Fishery resource utilization of a restored estuarine borrow pit: A beneficial use of dredged material case study. *Marine Pollution Bulletin*, 73, 115–128.

A before-and-after, site comparison study in 2004–2007 of two soft seabed sites in Barnegat Bay estuary, New Jersey, USA (1) found that partially refilling a disused borrow pit led to increased invertebrate species richness, abundance and diversity after 2–3 years, but these remained lower than at a nearby natural site. Refilling the pit increased average species richness (before: 1–13; after: 14–24 taxa/sample), abundance (before: 0–144; after: 151–495 individuals/sample) and diversity (presented as diversity indices) but these remained lower than at the natural site (species: 40; abundance: 1,370). Abundance at the natural site had increased over the same time (before: 435; after: 1,370) and species richness remained stable (before: 40; after: 40). In 2004, a borrow pit was partially filled with dredged sand, reducing its depth from 11.5 m to 6 m and increasing relief complexity-. Once in 2006 and twice in 2007, eighteen sediment samples were collected at the restored pit and six at a nearby natural site using a grab (0.044 cm^2, 6 cm depth). Invertebrates (>0.5 mm) were identified and counted. Data post-restoration (2006 and 2007) were pooled. Data prior to restoration were obtained from Versar (1999).

Versar (1999). *Biological sampling for dredged holes in Barnegat Bay*, Ocean County, NJ. Data Report prepared for the US Army Corps of Engineers, Philadelphia District, Philadelphia, PA.

(1) Reine K., Clarke D., Ray G. & Dickerson C. (2013) Fishery resource utilization of a restored estuarine borrow pit: A beneficial use of dredged material case study. *Marine Pollution Bulletin*, 73, 115–128.

12.6. Install a pump on or above the seabed in docks, ports, harbour, or other coastal areas to increase oxygen concentration

- **One study** examined the effects of installing a pump on or above the seabed in docks, ports, harbour, or other coastal areas to increase oxygen concentration on subtidal benthic invertebrate populations. The study was in Osaka Bay[1] (Japan).

COMMUNITY RESPONSE (1 STUDY)

- **Overall richness/diversity (1 study):** One before-and-after study in Osaka Bay[1] found that installing a pump on the seabed of a port to mix seawater and increase oxygen concentration led to an increase in combined invertebrate and fish species richness.

POPULATION RESPONSE (1 STUDY)

- **Overall abundance (1 study):** One before-and-after study in Osaka Bay[1] found that installing a pump on the seabed of a port to mix seawater and increase oxygen concentration led to an increase in combined invertebrates and fish abundance.

Background

Habitats and invertebrate populations within many ports, harbours, and docks around the world have deteriorated due to anthropogenic pressures, such as pollution and eutrophication, and the consequent decline in water quality and oxygenation (Russell *et al*. 1983). Improving water quality to these environments, for instance by increasing water mixing and oxygenation, can be achieved by installing a pump underwater. Installing a pump can help increase oxygen concentration in seawater, improve overall water quality, and potentially help promote subtidal benthic invertebrate biodiversity (Russell *et al*. 1983; Yamochi & Oda 2002).

Russell G., Hawkins S.J., Evans L.C., Jones H.D. & Holmes G.D. (1983) Restoration of a disused dock basin as a habitat for marine benthos and fish. *Journal of Applied Ecology*, 43–58.

Yamochi S. & Oda K. (2002) An attempt to restore suitable conditions for demersal fishes and crustaceans in the Port of Sakai-Semboku, north Osaka Bay, Japan. *Aquatic Ecology*, 36, 67–83.

A before-and-after study in 1995–1998 in one area of seabed in Osaka Bay, Japan (1) found that installing a pump on the seabed of a port to mix seawater and increase oxygen concentration appeared to increase combined invertebrate and fish species richness and abundance, after four months. Data were not statistically tested. Species richness was seven times higher after installing the pump (14 species/survey) compared to before (7 species/survey), and abundance was 52 times higher after (11 individuals/transect) than before (0.2 individuals/transect). In May 1996, a jet stream pump system was installed on the seabed of a port with low water oxygen concentration, at 4 m water depth. One dredge net (2 m x 0.5 m, 0.7–1.5 cm mesh size) was deployed along ten 70 m transects during weekly surveys before (June–August 1996; seven surveys) and after installation (June–August 1998; six surveys). Invertebrates and fish caught were identified and counted and results presented as combined species richness and abundance.

(1) Yamochi S. & Oda K. (2002) An attempt to restore suitable conditions for demersal fishes and crustaceans in the Port of Sakai-Semboku, north Osaka Bay, Japan. *Aquatic Ecology*, 36, 67–83.

Habitat enhancement

12.7. Landscape or artificially enhance the seabed (natural habitats)

- **Three studies** examined the effects of landscaping or artificially enhancing the seabed on subtidal benthic invertebrates. One study was in the North Sea[1] (UK), one in the Westerschelde estuary[2] (Netherlands), and one in the Persian Gulf[3] (Kuwait).

COMMUNITY RESPONSE (3 STUDIES)

- **Overall community composition (2 studies):** One controlled, before-and after study in the North Sea[1] found that following addition of gravels, invertebrate community composition became more similar to natural seabed communities. One before-and-after, site comparison study in the Westerschelde[2] estuary found no change in invertebrate community composition following addition of sedimentary dredge material.
- **Overall richness/diversity (3 studies):** One controlled, before-and after study in the North Sea[1] and one site comparison study in the Persian Gulf[3] found that invertebrate species richness increased following addition of gravels[1] or coral and limestone rubbles[3], and one also found that richness became similar to natural seabed[3]. One before-and-after, site comparison study in the Westerschelde[2] estuary found no change in species richness following addition of sedimentary dredged material.

POPULATION RESPONSE (3 STUDIES)

- **Overall abundance (3 studies):** One controlled, before-and after study in the North Sea[1] and one site comparison study in the Persian Gulf[3] found that invertebrate abundance and biomass increased following addition of gravels[1] or coral and limestone rubbles[3], and one also found that abundance became similar to natural seabed[3]. One before-and-after, site comparison study in the Westerschelde[2] estuary found no change in invertebrate abundance and biomass following addition of sedimentary dredge material.

Background

Landscaping or artificial enhancement can be undertaken as a restoration method aimed to promote the recovery and recolonization processes of subtidal benthic invertebrates. This could also constitute a pre-restoration method to improve the seabed and potentially enhance further restoration. . For instance, following cessation of marine aggregate extraction, to try recreating the natural substrate, a layer of gravel or shell can be added to the seabed or the sediment can be landscaped (Cooper *et al*. 2011; de Jong *et al.* 2015). Broken limestone rubble or empty shellfish shells have also been laid on the seabed to enhance the natural habitat and enhance recolonization of natural biodiversity (Fariñas-Franco *et al*. 2013; Jones & Nithyanandan 2013; Meyer & Townsend 2000).

Cooper K., Ware S., Vanstaen K. & Barry J. (2011) Gravel seeding - A suitable technique for restoring the seabed following marine aggregate dredging? *Estuarine, Coastal and Shelf Science,* 91, 121–132.

de Jong M.F., Baptist M.J., Lindeboom H.J. & Hoekstra P. (2015) Short-term impact of deep sand extraction and ecosystem-based landscaping on macrozoobenthos and sediment characteristics. *Marine Pollution Bulletin,* 97, 294–308.

Fariñas-Franco J.M., Allcock L., Smyth D. & Roberts D. (2013) Community convergence and recruitment of keystone species as performance indicators of artificial reefs. *Journal of Sea Research,* 78, 59–74.

Jones D.A. & Nithyanandan M. (2013) Recruitment of marine biota onto hard and soft artificially created subtidal habitats in Sabah Al-Ahmad Sea City, Kuwait. *Marine Pollution Bulletin*, 72, 351–356.

Meyer D.L. & Townsend E.C. (2000) Faunal utilization of created intertidal eastern oyster (*Crassostrea virginica*) reefs in the southeastern United States. *Estuaries*, 23, 34–45.

A controlled, before-and-after study in 2005–2007 in a sandy seabed area in the southern North Sea, UK (1) found that depositing gravels to recreate natural habitat after ceasing aggregate extraction changed invertebrate community composition and increased species richness, abundance and biomass, after 12 months. Community composition became less similar to that of a site without gravel and more similar to that of a natural site (similarity with site without gravel presented as graphical analyses; similarity with natural site community increased from 14% to 28%). Invertebrate species richness increased from 46/m^2 before gravel deposition to 118/m^2 after 12 months. There were also increases in invertebrate abundance (before: 222; after: 3,081 individuals/m^2) and biomass (before: 0.6; after: 7.5 g/m^2). In May 2005, July 2005, July 2006 and May 2007, invertebrates were surveyed at three sites at 22–33 m depths. Two sites were historically subjected to aggregate extraction (1996–2000), of which one was added 4,444 m^3 of gravels in July 2005 and the other left without gravel. The third site was natural (never subjected to aggregate extraction). Ten samples/site/survey were collected using sediment grabs (0.1 m^2). Invertebrates (>0.5 mm) were dried, weighed, and counted.

A before-and-after, site comparison study in 2004–2009 of two sites in one sandy seabed area in the Westerschelde estuary, southwestern Netherlands (2) found that disposing of dredge material in a shallow subtidal zone to enhance natural habitat did not affect invertebrate community composition, nor promote species richness, abundance, or biomass after up to five years. Invertebrate community composition did not change over time and remained different to that of the natural site (before: 65%; after: 31% similarity). After five years, average species richness (1.8 species/sample), abundance (data not reported) and biomass (5.4 mg/m^2) remained similar to pre-disposal values (species: 1.9; biomass: 7.2) and to values found at a nearby natural site (species: 1.7–1.8; biomass: 6.2–6.8). Dredged sand (500,000 m^3) was disposed at one site in November–December 2004. A second site (2 km away) was left natural. Yearly in spring and autumn between 2004 and 2009, three sediment cores (30 cm depth, 8 cm diameter) were taken (then pooled) at each of twenty locations/site. Invertebrates (> 1mm) were identified, counted, and dry-weighed.

A site comparison study in 2004–2011 in one soft seabed area in the Persian Gulf, southern Kuwait (3), found that seabed sites artificially enhanced by adding broken coral limestone rubble developed similar invertebrate species richness and abundance compared to natural sites, within two to six years. Results were not tested for statistical significance. Within two to six years, invertebrate species richness appeared similar in artificially enhanced (38.3–129 species/site) and natural site (66 species/site), and within one to two years abundance appeared higher in artificially enhanced (206–19,404 individual/m^3) than natural sites (2,263 individual/m^3). Following the construction of Sabah Al-Ahmad Sea City, waterways were gradually opened to the sea between 2004–2011 and sections of the seabed were artificially enhanced to promote colonisation. Annually in 2004–2011, samples were collected at 3–12 enhanced sites (3–4 m depth) using a sediment grab (three grabs/site). Invertebrates (>0.5 mm) were identified and counted. Data for the natural sites were obtained from surveys of 13 sites sampled in 2002–2004 by the United Nations Claim Commission (156 surveys in total, methodology not described).

(1) Cooper K., Ware S., Vanstaen K. & Barry J. (2011) Gravel seeding - A suitable technique for restoring the seabed following marine aggregate dredging? *Estuarine, Coastal and Shelf Science,* 91, 121–132.
(2) van der Wal D., Forster R.M., Rossi F., Hummel H., Ysebaert T., Roose F. & Herman P.M. (2011) Ecological evaluation of an experimental beneficial use scheme for dredged sediment disposal in shallow tidal waters. *Marine Pollution Bulletin,* 62, 99–108.
(3) Jones D.A. & Nithyanandan M. (2013) Recruitment of marine biota onto hard and soft artificially created subtidal habitats in Sabah Al-Ahmad Sea City, Kuwait. *Marine Pollution Bulletin,* 72, 351–356.

12.8. Use green engineering techniques on artificial structures

Background

Artificial structures are proliferating in the marine environment, with wide-scale negative impacts for biodiversity (Ido & Shimrit 2015). Green engineering, also referred to as “eco-engineering”, aims at mitigating the negative ecological impacts of artificial structures, and maximising their potential positive outcomes (Dafforn *et al*. 2015; Firth *et al*. 2014; Ido & Shimrit 2015). Green-engineering techniques include the use of natural or eco-friendly materials, such as wood, shell, rock, the addition of structural features (rockpools, crevices, ridges) or the seeding of native habitat-forming (biogenic) species, onto artificial structures (Dafforn *et al*. 2015; Firth *et al*. 2014).

Subsea cables and rock dump (used to stabilise and protect underwater structures, such as oil and gas platforms, as well as subsea cables and pipelines) can impact subtidal benthic invertebrates through physical damage, loss of natural sediment and changes in habitat characteristics. Modifying rock dump (either before or after deployment) to make it as similar as possible to natural hard substrates, such as bedrock or rocky reef, can potentially provide suitable replacement habitat for subtidal benthic invertebrate species commonly found in the area. In addition, subsea cables and pipelines could be covered with artificial reefs, or with materials that encourage the accumulation of natural sediment, to potentially promote colonisation by subtidal benthic invertebrates associated with natural hard rocky seabed or natural soft sediments, and enhance local biodiversity.

Evidence for other interventions related to rock dumping and offshore industries are summarised in the following chapters: "Threat: Energy production and mining" and "Threat: Transportation and service corridors". Evidence for other interventions related to subsea cables and pipelines are summarised under "Threat: Energy production and mining – Bury pipelines instead of surface laying and rock dumping", and "Threat: Transportation and service corridors – Bury cables and pipelines in the seabed rather than laying them on the seabed" and "Remove utility and service lines after decommissioning".

Dafforn K.A., Mayer-Pinto M., Morris R.L. & Waltham N.J. (2015) Application of management tools to integrate ecological principles with the design of marine infrastructure. *Journal of Environmental Management*, 158, 61–73.

Firth L.B., Thompson R.C., Bohn K., Abbiati M., Airoldi L., Bouma T.J., Bozzeda F., Ceccherelli V.U., Colangelo M.A., Evans A., Ferrario F., Hanley M.E., Hinz H., Hoggart S.P.G., Jackson J.E., Moore P., Morgan E.H., Perkol-Finkel S., Skov M.W., Strain E.M., van Belzen J. & Hawkins S.J. (2014) Between a rock and a hard place: Environmental and engineering considerations when designing coastal defence structures. *Coastal Engineering*, 87, 122–135.

Ido S. & Shimrit P.F. (2015) Blue is the new green–ecological enhancement of concrete based coastal and marine infrastructure. *Ecological Engineering*, 84, 260–272.

12.8.1. Modify rock dump to make it more similar to natural substrate

- We found no studies that evaluated the effects of modifying rock dump to make it more similar to natural substrate on subtidal benthic invertebrate 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.

12.8.2. Cover subsea cables with artificial reefs

- We found no studies that evaluated the effects of covering subsea cables with artificial reefs on subtidal benthic invertebrate 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.

12.8.3. Cover subsea cables with materials that encourage the accumulation of natural sediments

- We found no studies that evaluated the effects of covering subsea cables with materials that encourage the accumulation of natural sediments on subtidal benthic invertebrate 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.

12.9. Provide artificial shelters

- **Five studies** examined the effects of providing artificial shelters on subtidal benthic invertebrates. Three studies were in the Caribbean Sea[1,3,4] (Mexico); one in Florida Bay[2] and one in the Florida Keys[5] (USA).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (2 STUDIES)

- **Lobster abundance (2 studies):** Two replicated, controlled, before-and-after studies in the Caribbean Sea[1,4] found that abundance of lobsters either increased in plots with artificial shelters but not in plots without[1], or increased in all plots but more so in plots with artificial shelters than those without[4].
- **Lobster condition (1 study):** One replicated, controlled, before-and-after study in the Caribbean Sea[4] found that lobsters in plots with artificial shelters were bigger than in plots without.

BEHAVIOUR (3 STUDIES)

- **Use (3 studies):** Three replicated studies (two controlled) in Florida Bay[2], the Florida Keys[5], and the Caribbean Sea[3], found that artificial shelters were occupied by lobsters[2,3,5] and molluscs[2], that occupancy by lobsters varied with artificial shelter designs[2], that lobsters occupied artificial shelters more than natural ones (crevices)[3], and that lobsters occupying artificial shelters were larger, had greater nutritional condition, and had similar sex ratio and survival rate, compared to lobsters occupying natural shelters[5].

Background

Many populations of marine subtidal benthic invertebrate species have declined or been depleted due to the multiple threats they are under, such as habitat loss (Airoldi *et al*. 2008). To improve animals' survival or recolonization success, some assistance can be provided in the form of artificial shelters that mimic the natural shelters usually found in the marine environment, such as 'casitas' used for lobsters (Gutzler *et al*. 2015). These shelters can provide refugia and potentially help animals avoid predation (Arce *et al.* 1997).

Airoldi L., Balata D. & Beck M.W. (2008) The gray zone: relationships between habitat loss and marine diversity and their applications in conservation. *Journal of Experimental Marine Biology and Ecology*, 366, 8–15.

Arce A.M., Aguilar-Dávila W., Sosa-Cordero E. & Caddy J.F. (1997) Artificial shelters (casitas) as habitats for juvenile spiny lobsters *Panulirus argus* in the Mexican Caribbean. *Marine Ecology Progress Series*, 158, 217–224.

Gutzler B.C. Butler M.J. & Behringer D.C. (2015) Casitas: A location-dependent ecological trap for juvenile Caribbean spiny lobsters, *Panulirus argus*. *ICES Journal of Marine Science,* 72, 177–184.

A replicated, controlled, before-and-after study in 1997–1999 of nine plots in a reef lagoon with seagrass meadows in the Caribbean Sea, Mexico (1 – same experimental setup as 4) found that during one year after deploying artificial shelters ('casitas'), abundance of Caribbean spiny lobsters *Panulirus argus* increased in plots with artificial shelters but not in plots without, regardless of whether the plots had lobsters or not before deployment. Lobster abundance increased over time in plots with shelters (before deployment: 0–30; after: 7–104 lobsters/ha) while it did not increase in plots without (before: 0–24; after: 0–32 lobsters/ha). After a year, plots with artificial shelters had bigger lobsters (average 25 mm) than sites without (22 mm). Ten 'casitas' (1.1 m^2, 3.8 cm diameter entrance) were deployed in each of five plots (1 ha) (≤5 m depth) in July 1998. On 13 occasions before (February 1997–July 1998) and every two months for a year after deployment, divers counted and visually estimated the carapace length of all lobsters across each 'casitas' plots and at four plots without casitas. Before deployment, three 'casitas' plots and two plots without 'casitas' had zero lobsters. No fishing is reported to occur in the study area because all lobsters present are typically undersize juveniles.

A replicated, controlled study in 1993–1995 in four sites across two areas of seagrass in the Everglades National Park, Florida Bay, USA (2) found that providing artificial shelters had varied effects on Caribbean spiny lobster *Panulirus argus* abundance (shelter occupancy) depending on shelter designs, but all designs had similar abundance of their molluscan preys. Lobster abundance was greater in full-roof artificial shelters (13 lobsters/shelter) than mesh-roof shelters (6 lobsters/shelter), and more lobsters were found in these two designs than in either artificial shelter frames (1 lobster/shelter) or plots without artificial shelters (0 lobster/plot). Abundance was not significantly different in shelter frames and plots without shelters. Abundance data for molluscs in each treatment were not shown. Four treatments (three shelter designs and no shelter) were tested; 1) an artificial shelter frame without a roof, 2) an artificial shelter frame covered with a 3.8 cm diamond mesh roof, 3) an artificial shelter frame covered with an aluminium sheet and 4) delimited empty plot of similar size as an artificial shelter. In July 1993, at two sites (1 km apart) in each of two areas (12 km apart), artificial shelters were deployed (4 replicates/treatment/site). Quarterly in 1993–1994 and twice in 1995, divers counted and measured lobsters >20 mm, and using a suction sampler identified and counted molluscs >1 mm sampled (1 replicate/treatment/sampling time; 0.05 m^2 samples) in all treatments.

A replicated study in 1998–2002 of nine plots in a reef lagoon with seagrass meadows in the Caribbean Sea, Mexico (3) found that over the 2.5 years after their deployment, artificial shelters ('casitas') were occupied by more and bigger Caribbean spiny lobsters *Panulirus argus* than natural shelters (crevices). More lobsters were found occupying artificial shelters (3.5–5.5 on average; 3,707 in total) than crevices (1–1.6 on average; 200 in total). In addition, artificial shelters hosted larger lobsters (30–32 mm carapace length) than crevices (17–18 mm). Ten 'casitas' (1.1 m^2, 3.8 cm diameter entrance) were deployed in each of five plots (1 ha) (≤5 m depth) in July 1998. On 22 occasions after deployment (September 1998–November 2002), divers counted and visually estimated the carapace length of all lobsters inside all artificial shelters and inside all naturally occurring shelters (crevices) in each plot.

A replicated, controlled, before-and-after study in 1997–2002 of nine plots in a reef lagoon with seagrass meadows in the Caribbean Sea, Mexico (4 – same experimental setup as 1) found that over the 2.5 years after deploying artificial shelters ('casitas'), abundance of Caribbean spiny lobsters *Panulirus argus* increased in all plots but more so in plots with artificial shelters than those without, and that lobsters in plots with artificial shelters were bigger. Before deployment, all plots had similar lobster abundance (1.5–8 lobsters/ha). After deployment, abundance increased approximately four times more in plots with artificial shelters (17–83 lobsters/ha) than plots without (5–21 lobsters/ha). Plots with artificial shelters had bigger lobsters (31 mm) than sites without (24 mm). Ten 'casitas' (1.1 m^2, 3.8 cm diameter entrance) were deployed in each of five plots (1 ha) (≤5 m depth) in July 1998. On six occasions before (December 1997–July 1998) and 22 occasions after (September 1998–November 2002) deployment, divers counted all lobsters) across each 'casitas' plots and at four plots without casitas. For the 'after' surveys only, the carapace length of lobsters was also visually estimated.

A replicated, controlled study in summer 2012–2013 of multiple sites in two areas of either rocky, sandy or seagrass bed in the Florida Keys, USA (5) found that the effects of artificial shelters ('casitas') on the nutritional condition of Caribbean spiny lobsters *Panulirus argus* varied with location, and that lobsters occupying them had similar sex ratio and typically similar survival, compared to lobsters in natural shelters, but lobsters in the artificial shelters were larger. The nutritional condition of lobsters (data presented as an index) differed between the two areas, but within each area lobsters inside artificial shelters had greater nutritional condition than lobsters in crevices. In addition, artificial shelters hosted larger lobsters (average 66 mm) than crevices (52 mm). Each artificial shelter was on averaged occupied by 22–41 lobsters. In three of four comparisons, lobster survival following predation experiment was similar in artificial (59–97% survival) and natural shelters (57–93% survival). In one comparison (of lobsters <35 mm), survival was lower in artificial (56% survival), compared to natural shelters (82% survival). The 'casitas' (4 m^2) were flat rectangular structures with at least two open sides. In one area, 'casitas' (number unspecified) had been deployed in 1990 (at 2–3 m depth). In the other area, 16 were deployed (at 10 m depth). In May–August 2012–2013, divers counted lobsters occupying 16 of the 'casitas', and collected lobsters using hand nets and tail snares found inside artificial and natural shelters at all sites. All lobsters were measured (carapace length), their sex recorded, and a subset was used to assess nutritional condition based on the weight of their lobster's digestive gland. Divers also experimentally assessed lobster mortality from predation inside artificial and natural shelters (using a tethering method).

(1) Briones-Fourzàn P. & Lozano-Álvarez E. (2001) Effects of artificial shelters (Casitas) on the abundance and biomass of juvenile spiny lobsters *Panulirus argus* in a habitat-limited tropical reef lagoon. *Marine Ecology Progress Series,* 221, 221–232.
(2) Nizinski M.S. (2007) Predation in subtropical soft-bottom systems: Spiny lobster and molluscs in Florida Bay. *Marine Ecology Progress Series,* 345, 185–197.
(3) Lozano-Álvarez E., Meiners C. & Briones-Fourzán P. (2009) Ontogenetic habitat shifts affect performance of artificial shelters for Caribbean spiny lobsters. *Marine Ecology Progress Series*, 396, 85–97.
(4) Lozano-Álvarez E., Briones-Fourzán P., Álvarez-Filip L., Weiss H.M., Negrete-Soto F. & Barradas-Ortiz C. (2010) Influence of shelter availability on interactions between Caribbean spiny lobsters and moray eels: Implications for artificial lobster enhancement. *Marine Ecology Progress Series,* 400, 175–185.
(5) Gutzler B.C. Butler M.J. & Behringer D.C. (2015) Casitas: A location-dependent ecological trap for juvenile Caribbean spiny lobsters, *Panulirus argus*. *ICES Journal of Marine Science,* 72, 177–184.

Artificial habitat creation

12.10. Create artificial reefs

- **Twelve studies** examined the effects of creating artificial reefs on subtidal benthic invertebrate populations. Three studies were in the Mediterranean Sea[1,2,3] (Italy); three were in the North Atlantic Ocean[4,8,11] (USA, Portugal, France); one in the Firth of Lorn[6] (UK); two in the North Pacific Ocean[5,9] (USA); one in the English Channel[7] (UK), one in the Gulf of Mexico[10] (USA); and one in the Yellow Sea[12] (China).

COMMUNITY RESPONSE (8 STUDIES)

- **Overall community composition (3 studies):** Two site comparison studies (one replicated) in the English Channel[7] and North Atlantic Ocean[8] found that invertebrate communities growing on artificial reefs were different to that of natural reefs. One replicated study the North Pacific Ocean[9] found that invertebrate community composition changed over time on an artificial reef.
- **Overall richness/diversity (6 studies):** Two site comparison studies (one replicated) in the Mediterranean Sea[3] and North Atlantic Ocean[8] found that invertebrate species richness and/or diversity on the artificial reef[8] or in the sediments[3] inside and adjacent to the reef area were lower compared to on natural reefs or in nearby natural sediments. One replicated, site comparison study in the Gulf of Mexico[10] found that artificial breakwaters had more species of nekton compared to adjacent mudflats. One site comparison study in English Channel[7] recorded 263 taxa on the artificial reef, including at least nine not recorded on nearby natural reefs but excluding at least 39 recorded on natural reefs. One replicated study in the North Pacific Ocean[9] found a 49% increase in species richness over five years on an artificial reef. One study in the North Atlantic Ocean[11] found that artificial reefs hosted at least five species of large mobile invertebrates.
- **Mollusc richness/diversity (1 study):** One replicated, site comparison study in the Mediterranean Sea[1] found that mollusc species richness and diversity were lower on artificial reefs compared to natural reefs.
- **Worm community composition (1 study):** One replicated, site comparison study in the North Pacific Ocean[5] found that polychaete worm community composition was similar at one of two artificial reefs compared to a natural reef.
- **Worm richness/diversity (1 study):** One replicated, site comparison study in the North Pacific Ocean[5] found that polychaete worm species richness and diversity were similar at one of two artificial reefs compared to a natural reef, but lower at the second artificial reef.

POPULATION RESPONSE (12 STUDIES)

- **Overall abundance (10 studies):** One of two site comparison studies (one replicated) in the Mediterranean Sea[2,3] found that abundance of invertebrates in the sediment was lower at the reef sites than in nearby natural sediments, but increased in the sediments directly adjacent to the reefs, while the other study found that abundance was similar in the sediments inside and directly adjacent to the artificial reef area, but lower than in nearby natural sediments. Of five site comparison studies (four replicated) in the North Pacific Ocean[5], the North Atlantic Ocean[4,8], the Gulf of Mexico[10] and the Yellow Sea[12], one[4] found that invertebrate biomass was higher on the artificial reef than in adjacent natural sediments, two that invertebrate abundance and biomass[8] and nekton abundance[10] were similar on artificial reefs and natural habitats (reef[8]; mudflat[10]), and two[5,12] found mixed effects on abundances of invertebrates. One site comparison study in the English Channel[7] reported that the abundances of some species were lower on the artificial reef compared to natural reefs. One replicated study in the North Pacific Ocean[9] reported an 86% increase in invertebrate abundance growing on an artificial reef o over five years. One study in the North Atlantic Ocean[11] found that two of five species at one artificial reef, and three of seven at another, were recorded during >50% of dives.
- **Overall condition (1 study):** One replicated, site comparison study in the Yellow Sea[12] found mixed effects of creating an artificial reef on the sizes of mobile invertebrates.

- **Mollusc abundance (1 study):** One replicated, site comparison study in the Mediterranean Sea[1] found that mollusc abundance was lower on artificial reefs compared to natural reefs.
- **Crustacean abundance (1 study):** One replicated, site comparison in the Firth of Lorn[6] found that abundances of edible crabs and velvet swimming crabs were typically higher on artificial than natural reefs.

OTHER (1 STUDY)

- **Biological production (1 study):** One site comparison study in North Atlantic Ocean[4] found that secondary production was higher from invertebrates growing on an artificial reef than from invertebrates in adjacent natural sediments.

Background

Artificial reefs are man-made structures intentionally introduced into the marine environment and aimed to act similarly to natural reefs. Originally used to improve biological resources, they may also increase other local biodiversity (Bohnsack & Sutherland 1985; Clark & Edwards 1999). Creating an artificial reef in an area can potentially help increase subtidal benthic invertebrate biodiversity by creating additional suitable habitat for them.

Evidence comparing artificial reefs of different typologies (material used and/or 3-D structures) is summarised under "Habitat restoration and creation – Create artificial reefs of different 3-D structure and material used". When artificial reefs are created as an offset strategy, evidence has been summarised under "Habitat restoration and creation – Offset habitat loss from human activity by restoring or creating habitats elsewhere".

Bohnsack J.A. & Sutherland D.L. (1985) Artificial reef research: a review with recommendations for future priorities. *Bulletin of Marine Science*, 37, 11–39.

Clark S. & Edwards A.J. (1999) An evaluation of artificial reef structures as tools for marine habitat rehabilitation in the Maldives. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 9, 5–21.

A replicated, site comparison study in 1995 of three artificial and two natural reefs in the Mediterranean Sea, off the coast of northwest Sicily, Italy (1) found that artificial reefs developed similar molluscan abundance but not species richness or diversity to natural reefs after three years. Abundance was similar on artificial reefs (41–50 individuals/sample) and natural reefs (abundance: 19–42 individuals/sample). However, molluscan species richness and diversity (as diversity indices) were lower on artificial reefs (4–11 species/sample) compared to natural reefs (10–27 species/sample). Of the 166 species found in total across all reefs, only 29% were found on both artificial and natural reefs. In spring 1995, molluscs were surveyed on three artificial reefs made of concrete created three years earlier and two nearby natural reefs (0.5–4.5 km from the artificial reefs). A total of 28 samples (400 cm^2 each) were manually collected at 16–22 m depth (4–8/artificial reef; 4/natural reef). All molluscs were identified and counted.

A replicated, site comparison study in 1997–1998 of sandy sediments surrounding two artificial reefs in the Mediterranean Sea, off the coast of Italy (2) found that the effects of creating artificial reefs on small invertebrate abundance varied with distance to the reefs. Abundance was lower at the artificial reef sites (87–180 individual/10 cm^2) compared to nearby natural sites (146–265 individual/10 cm^2). However, abundance was higher at sites adjacent to the artificial reefs 2–20 m away (135–332 individual/10 cm^2). Authors suggested that the lower abundance at the artificial reef sites was linked with a higher silt-clay:sand ratio and changes in oxygen penetration. In winter

1997/1998 and summer 1998, small invertebrates were surveyed in the sediments surrounding two artificial reefs (groups of pyramids; material unspecified). One was created in 1987, and the other in 1992. Samples were taken with increasing distance from the reef: one at 0 m (artificial reef site), three at 2–20 m (affected adjacent sites), and one at 50 m (unaffected natural site). Sediment samples (3/station) were collected using a core (4.6 cm diameter, 10 cm depth), and invertebrates (37 μm–1 mm) identified and counted.

A site comparison study in 1997–1999 of sandy sediments surrounding an artificial reef and at a natural site in the Mediterranean Sea, off the coast of Italy (3) found that invertebrate species richness and abundance tended to be similar in the sediments inside the artificial reef area and directly adjacent to it, but lower than at a nearby natural site. Data were not statistically tested. Total invertebrate species richness was 91–109 species/station inside the reef area, 79–88/station 2–20 m away, 92/station 50 m away, and 96/station at the natural site. Average invertebrate abundance was 930–1,000 individuals/m^2 inside the reef area, 750–930/m^2 2–20 m away, 1,010/m^2 50 m away, and 2,060/m^2 at the natural site. An artificial reef made of 29 concrete pyramids was created in 1987. Seasonally in 1997–1999, invertebrates were surveyed in the sediments at 17 stations: six within the reef area, eight 2–20 m from the edge of the reef, two 50 m from the reef, and one at a natural site (2.5 nm away). Invertebrates (>0.5 mm) were sampled using a suction sampler from 1,600 cm^2 quadrats (3 quadrats/station/survey), identified and counted.

A site comparison study in 1990–1994 of an artificial reef and nearby sandy habitat in Delaware Bay, North Atlantic Ocean, USA (4) found that invertebrate biomass and secondary production were higher on the artificial reef than in adjacent natural sediments over five years. Average invertebrate biomass was higher on the reef (8,000 g/m^2) than in the nearby sediments (180 g/m^2). Average estimated secondary production (measure of consumers biomass regeneration over time) was also higher from invertebrates growing on the reef (3,990–9,555 kcal/m^2/year) compared to invertebrates in the sediments (215–249 kcal/m^2/year). This corresponded to an increase in average secondary productivity by a factor of 19–38 on artificial reef habitat compared to natural sandy habitat. An artificial reef made of complex concrete panels was created in 1989 to mitigate the loss of mudflats elsewhere. Twice per summer in 1990–1994, sessile invertebrates (>0.05 mm) growing on the artificial reef and within nearby sediments were identified and their biomass measured. Biomass data were used to estimate annual secondary production.

A replicated, site comparison study in 2005 of three reefs in Malaya Bay, Hawai'i, North Pacific Ocean, USA (5) found that overall invertebrate abundance was similar at one but lower at a second artificial reef, compared to a natural reef. Average invertebrate abundance was similar at the sunken vessel *Sea Tiger* (131 individuals/sample) and at the natural reef (115), but lower at the sunken vessel *YO257* (47). In addition, polychaete worm (the dominant group at all sites) diversity (reported as a diversity index) and species richness were similar at *Sea Tiger* (16 species) and the natural reef (13), but were lower at *YO257* (8.3). Polychaete community composition was similar between *YO257* and the natural site, but significantly different at *Sea Tiger* thought to be due to the development of seagrass (data presented as graphical analysis and statistical model results). Two vessels were deployed as artificial reefs on sandy seabed 1.5–2 km off the coast at 35–38 m water depth: the *YO257* in 1989 (along with some gravels) and the *Sea Tiger* in 1999. Two transect lines were surveyed at each artificial reef (one on each side), and one at a natural reef located 1.5 km off the coast at 32 m depth. Divers collected six sediment samples/transect by randomly placing corers (7.6 cm diameter, 6 cm depth). Invertebrates (>500 μm) were identified and counted.

A replicated, site comparison study in 2005–2006 of nine sites in the Firth (Lynn) of Lorn, west coast of Scotland, UK (6) found that abundances of edible crab *Cancer pagurus* and velvet swimming crab *Necora puber* were typically higher on artificial than natural reefs, but varied with the complexity of the reefs and the season. For edible crabs, in summer and autumn abundances were similar at artificial and natural reefs and averaged 0–0.05/m^2. In winter, abundance was higher at one of two types of artificial reefs (0.13/m^2), compared to natural reefs (0.01/m^2), but not in the other artificial reef type (0.04/m^2). In spring, abundance was not significantly different at artificial and natural reefs and averaged 0.04–0.15/m^2. For swimming crabs, in summer abundance was higher at artificial reefs (0.15–0.27/m^2) than at natural reefs (0.08/m^2). In all other seasons, abundance was higher at one of two types of artificial reefs (0.34–0.45/m^2), than natural reefs (0.10–0.14/m^2), but not in the other artificial reef type (0.05–0.18/m^2). In 2003–2004, an artificial reef complex made of two types of modules (concrete blocks; perforated concrete blocks) was created. Nine sites were surveyed: six with artificial modules and three nearby natural reefs. Monthly in August 2005–June 2006, divers recorded edible and swimming crab abundance along two 9 m^2 belt transect/site. Data were grouped by season.

A site comparison study in 2004–2009 in two areas off the coast of south Cornwall and Devon, English Channel, UK (7) found that the invertebrate and algae community found on an artificial reef was different to that of nearby natural reefs five years after its creation. Results were not statistically tested. After five years, 263 taxa were found on the artificial reef. The total number of taxa on natural reefs was not specified. Nine conspicuous species were only found on the artificial reef, and 39 conspicuous species were only found on the natural reefs (definition of "conspicuous" unspecified). The abundance of some species on the artificial reef was reported to be lower compared to natural reefs (see paper for details). An ex-Royal Navy boat was placed on the seabed for recreational purposes in March 2004 at 20 m depth. The occurrence and abundance of invertebrates and algae were recorded by divers opportunistically in 2004–2009 (approximately monthly in the first 18 months and then approximately every 10 weeks). Divers also took photographs. The invertebrate and algae community present end of summer 2008 was compared to that of nearby natural bedrock reefs previously surveyed (number of sites unspecified).

A replicated, site comparison study in 2006 of two artificial and two natural reefs in the Faro/Ancão reef system, off the southern coast of Portugal, North Atlantic Ocean (8) found that artificial reefs developed similar invertebrate abundance and biomass, but not similar invertebrate species richness, diversity and community composition to natural reefs after 16 years. Invertebrate abundance and biomass were similar on artificial reefs (abundance: 17,111–52,933 individual/m^2; biomass: 18–40 g/m^2) and natural reefs (abundance: 16,400–25,644 individual/m^2; biomass: 27–262 g/m^2). However, species diversity (as diversity index) and richness were lower on artificial reefs (162 species) compared to natural reefs (218 species). Invertebrate community composition was different on artificial reefs compared to natural reefs (data presented as graphical analyses and statistical model results). In August 2006, two artificial reefs made of concrete created in 1990 and two natural reefs (0.5–0.9 km away from the artificial reefs) were surveyed. Three 15 x 15 cm quadrats were placed at each reef on vertical surfaces 1 m from the seabed, scraped, and organisms collected. Invertebrates (>0.5 mm) and algae were identified, counted, and dry-weighed.

A replicated study in 1999-2004 of one artificial reef off southern California, North Pacific Ocean, USA (9) found that from one to five years after artificial reef modules were deployed there were changes in community composition, and an increase in species richness (by 49%) and abundance (by 86%) of invertebrates growing on the reef modules (sessile). Over time, artificial reef modules had increased invertebrate species richness (2000: 4 species/m^2, 2004: 7 species/m^2) and abundance (2000: 47% cover, 2004: 70%). The artificial reef was created to compensate for the loss of giant kelp forest. Low lying (<1 m tall) artificial reef modules (40 x 40 m) made of either granite boulders or concrete rubble were deployed in seven sites in 1999 (8 modules/site) at 13–16 m depth. Sessile invertebrate communities were sampled in summer one and five years after deployment. Invertebrate abundance was assessed for 42 of the 56 modules using six 1 m^2 quadrats/modules.

A replicated, site comparison study in 2008–2009 of six sites in northwest Mobile Bay, Gulf of Mexico, Alabama, USA (10) found that artificial breakwaters had more small mobile animal species (invertebrates and fish combined, referred to as "nekton"), but similar overall nekton abundance compared to adjacent mudflats 1.5 years after deployment. Artificial breakwaters had more species of nekton (2.2–2.3 species/m^2) compared to adjacent mudflats (1.3 species/m^2). However, breakwaters did not have statistically higher nekton abundance (0.5 individual/m^2) compared to mudflats (0.1 individual/m^2). Four artificial breakwaters made of either bagged oyster shells or concrete domes, acting as artificial reefs, were deployed in May 2008 along an eroding shoreline in Mobile Bay (60 m from, and parallel to the shore; 0.75 m depth). Between May 2008 and November 2009, nekton was surveyed at each breakwater and at two adjacent natural mudflats. During each survey, a bag seine (6.25 mm mesh) was deployed over 12.5 m on each side of the breakwaters and twice in the mudflats. All individuals were identified and counted.

A study in 2009–2013 of two artificial reefs in the southern Bay of Biscay, North Atlantic Ocean, France (11) found that artificial reefs hosted at least five to seven species of large mobile invertebrates. Five species were recorded in Porto artificial reef, with two recorded during >75% of dives (edible crabs *Cancer pagurus*; velvet crabs *Necora puber*). Seven species were recorded in Capbreton artificial reef, with one recorded during >75% of dives (the common octopus *Octopus vulgaris*) and two recorded during 50–75% of dives (the common prawn *Palaemon serratus*; the velvet crab). Other large mobile invertebrate species recorded in lower frequencies included European spider crabs *Maja brachydactyla*, hermit crabs *Pagurus bernhardus*, and common cuttlefish *Sepia officinallis*. Porto artificial reef was created in 1994 and made more complex over time until 2004. Capbreton artificial reef was created in 1999 and made more complex in 2010. Both reefs were made of barges, concrete modules and pipes, and were located on sandy seabed at 12–25 m depth, 84 km and 20 km away from the nearest rocky shore, respectively. Anchoring, diving, and all types of fishing were prohibited. Annually between 2009–2012

(Porto) and 2010–2013 (Capbreton), 2–4 stations/artificial reef were surveyed during 2–5 dives/station. During each dive, two divers visually recorded and counted the number of large mobile invertebrate species in a 2 m radius circle for 3 min. Frequency of occurrence was calculated for each species as: (number of dives in which the species was counted/total number of dives) X 100.

A replicated, site comparison study in 2012–2013 of 29 sites in three areas of Shandong province, Yellow Sea, China (12) found that creating artificial reefs had mixed effects on the abundances and sizes of mobile invertebrates. Of 17 species found at both artificial reefs and natural sites with no artificial reefs, abundances tended to be higher at artificial reef sites compared to natural sites for 10 species, lower for six, and unrecorded for one (see original paper for details). Individual sizes tended to be higher at artificial reef sites compared to natural sites for seven species, equal for one, lower for six, and unrecorded for three. Differences were not statistically tested. Artificial reefs made of various materials and structures (including natural rock, stones, concrete blocks, concrete pipes, concrete slaps, and wooden shipwrecks) were created in 2005–2010 to boost fisheries. Three areas were chosen, and 3–8 artificial reef sites selected/area. For comparison, 3–6 natural sites/area were also selected located 800 m from the artificial reefs. During five surveys between September 2012 and August 2013, mobile invertebrates were sampled at each site (but not directly on the artificial reefs) using nets (28 m long, 3 m high, 10 cm outer mesh, 4 cm inner mesh) soaked for 24h. Invertebrates were identified, counted, and measured.

(1) Badalamenti F., Chemello R., D'anna G., Ramos P.H. & Riggio S. (2002) Are artificial reefs comparable to neighbouring natural rocky areas? A mollusc case study in the Gulf of Castellammare (NW Sicily). *ICES Journal of Marine Science*, 59, 127–131.

(2) Danovaro R., Gambi C., Mazzola A. & Mirto S. (2002) Influence of artificial reefs on the surrounding infauna: analysis of meiofauna. *ICES Journal of Marine Science*, 59, 356–362.

(3) Fabi G., Luccarini F., Panfili M., Solustri C. & Spagnolo A. (2002) Effects of an artificial reef on the surrounding soft-bottom community (central Adriatic Sea). *ICES Journal of Marine Science*, 59, 343–349.

(4) Steimle F., Foster K., Kropp R. & Conlin B. (2002) Benthic macrofauna productivity enhancement by an artificial reef in Delaware Bay, USA. *ICES Journal of Marine Science*, 59, 100–105.

(5) Fukunaga A. & Bailey-Brock J.H. (2008) Benthic infaunal communities around two artificial reefs in Mamala Bay, Oahu, Hawaii. *Marine Environmental Research*, 65, 250–263.

(6) Hunter W.R. & Sayer M.D.J. (2009) The comparative effects of habitat complexity on faunal assemblages of northern temperate artificial and natural reefs. *ICES Journal of Marine Science*, 66, 691–698.

(7) Hiscock K., Sharrock S., Highfield J. & Snelling D. (2010) Colonization of an artificial reef in south-west England ex-HMS Scylla. *Journal of the Marine Biological Association of the United Kingdom,* 90, 69–94.

(8) Carvalho S., Moura A., Cúrdia J., da Fonseca L.C. & Santos M.N. (2013) How complementary are epibenthic assemblages in artificial and nearby natural rocky reefs? *Marine Environmental Research*, 92, 170–177.

(9) Schroeter S.C., Reed D.C. & Raimondi P.T. (2015) Effects of reef physical structure on development of benthic reef community: A large-scale artificial reef experiment. *Marine Ecology Progress Series,* 540, 43–55.

(10) Scyphers S.B., Powers S.P. & Heck K.L. (2015) Ecological value of submerged breakwaters for habitat enhancement on a residential scale. *Environmental Management*, 55, 383–391.

(11) Castège I., Milon E., Fourneau G. & Tauzia A. (2016) First results of fauna community structure and dynamics on two artificial reefs in the south of the Bay of Biscay (France). *Estuarine, Coastal and Shelf Science*, 179, 172–180.

(12) Sun P., Liu X., Tang Y., Cheng W., Sun R., Wang X., Wan R., Heino M. & Handling editor: Jonathan Grabowski (2017) The bio-economic effects of artificial reefs: mixed evidence from Shandong, China. *ICES Journal of Marine Science*, 74, 2239–2248.

12.11. Create artificial reefs of different 3-D structure and material used

- **Eight studies** examined the effects of creating artificial reefs of different typology on subtidal benthic invertebrate populations. One study was in the English Channel[1] (UK), three in the Mediterranean Sea[2,5,6] (Israel, Italy), one in the North Atlantic Ocean[3] (USA), one in the Firth of Lorn[4] (UK), one in the North Pacific Ocean[7] (USA), and one in the Gulf of Mexico[8] (USA).

COMMUNITY RESPONSE (6 STUDIES)

- **Overall community composition (3 studies):** One controlled study in the English Channel[1] found that artificial reef modules made of scrap tyres developed a similar sessile invertebrate community composition as traditional artificial concrete modules. Two controlled studies (one replicated) in the Mediterranean Sea[5,6] found that pyramid reefs made of "sea-friendly" concrete developed different invertebrate community compositions compared to reefs of either traditional concrete plinth-pole structures[5] or bundles of traditional concrete tubes[6].
- **Overall richness/diversity (5 studies):** Four controlled studies (three replicated) in the Mediterranean Sea[2,5], the North Pacific Ocean[7], and the Gulf of Mexico[8] found no differences in overall invertebrate richness/diversity[2,5,7] or combined mobile invertebrate and fish richness[8] between reef structure and/or material. One controlled study in the Mediterranean Sea[6] found that invertebrate species richness was lower on "sea-friendly" pyramid reefs compared to bundle reefs of traditional concrete.

POPULATION RESPONSE (7 STUDIES)

- **Overall abundance (5 studies):** Four controlled studies (three replicated) in the English Channel[1], the Mediterranean Sea[2], the North Pacific Ocean[7], and the Gulf of Mexico[8] found no differences in overall invertebrate abundances[1,2,7] or combined mobile invertebrate and fish abundance[8] between reef structure and/or material. One controlled study in the Mediterranean Sea[5] found that "sea-friendly" concrete pyramids had lower abundance compared to plinth-pole structures after two years, but higher after three.
- **Crustacean abundance (2 studies):** One replicated, controlled study in the North Atlantic Ocean[3] found that artificial reefs made of limestone boulders, gravel concrete aggregate, or tyre-concrete aggregate had similar abundance of spiny lobsters. One replicated, controlled study in the Firth of Lorn[4] found that the complexity of artificial reef modules had mixed effects on the abundance of edible crab and velvet swimming crab.
- **Mollusc abundance (1 study):** One replicated, controlled study in the Gulf of Mexico[8] found that breakwaters made of bags of oyster shells recruited more oysters and ribbed mussels compared to "ReefBall" breakwaters.

Background

Artificial reefs are man-made structures intentionally put into the marine environment to act similarly to a natural reef. Originally used to improve fisheries and biological resources, they have been shown to be ecologically beneficial by locally increasing biodiversity (Bohnsack & Sutherland 1985; Clark & Edwards 1999). Various construction material and architectural arrangements can be used when creating an artificial reef to manipulate its 3-D structure and level of complexity, for instance by using pyramidal structures instead of tubes, or using "sea-friendly" reinforced concrete instead of traditional common concrete (Ponti *et al.* 2015; Spagnolo *et al.* 2014). The architecture and characteristics of the artificial reef can influence which species of subtidal benthic invertebrates colonize the reefs, in what abundances, at what rate, and how long they can survive. The 3-D structure and material used to create an artificial reef can be selected to potentially enhance marine subtidal biodiversity.

Evidence related to creating artificial reefs in general, without considering 3-D structures or construction material, is summarised under "Habitat restoration and creation – Create artificial reefs".

Bohnsack J.A. & Sutherland D.L. (1985) Artificial reef research: a review with recommendations for future priorities. *Bulletin of Marine Science*, 37, 11–39.

Clark S. & Edwards A.J. (1999) An evaluation of artificial reef structures as tools for marine habitat rehabilitation in the Maldives. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 9, 5–21.

Ponti M., Fava F., Perlini R.A., Giovanardi O. & Abbiati M. (2015) Benthic assemblages on artificial reefs in the northwestern Adriatic Sea: Does structure type and age matter? *Marine Environmental Research*, 104, 10–19.

Spagnolo A., Cuicchi C., Punzo E., Santelli A., Scarcella G. & Fabi G. (2014) Patterns of colonization and succession of benthic assemblages in two artificial substrates. *Journal of Sea Research*, 88, 78–86.

A controlled study in 1998–1999 of an unclear number of artificial reef modules in Poole Bay, English Channel, UK (1) found that modules made of scrap tyres developed a similar sessile invertebrate community composition and species percentage cover compared to traditional artificial concrete modules, 10–11 months after deployment. Tetrahedral and cylindrical tyre modules had similar community composition to concrete modules (community data presented as graphical analyses). Tetrahedral and cylindrical tyre modules also had similar species groups percentage cover compared to concrete modules (see paper for specific groups). In July 1998, artificial modules (number unclear) arranged in eight groups were put on the seabed as artificial reefs alongside a pre-existing coal ash artificial reef. Each group had replicate modules of each of three reef types: tetrahedral tyre lattice (4–13 tyres/module), cylindrical tyre stack (6–7 tyres/module), or traditional concrete block (no tyres). Every two month until June 1999, divers photographed the sides of the modules (8 photographs/module), and invertebrate growing on them were identified, and their percentage cover assessed.

A replicated, controlled study in 1996–1999 of 80 artificial reef blocks in the southeastern Mediterranean Sea, off the coast of Haifa, Israel (2) found that blocks created using coal fly ash (a cheap waste product) instead of sand had similar sessile invertebrate species richness and percentage cover as traditional blocks without coal fly ash, over 33 months following deployment. When compared to blocks made of traditional 0% coal fly ash, reef blocks made of either 40%, 60% or 80% coal fly ash had similar species richness (in 22 of 24 comparisons) and similar species cover (in 24 of 24 comparisons) (data not shown). In November 1996, blocks (20 × 20 × 40 cm) made of a mixture of concrete and coal fly ash were put on the seabed as artificial reefs at 18.5 m depth. There were four treatments: blocks with either 0%, 40%, 60% and 80% coal fly ash (20 blocks/treatment). Divers sampled two blocks/treatment at 3–4-month intervals for 33 months (10 sampling events). Invertebrates growing on each block side were identified and counted, and their percentage cover estimated.

A replicated, controlled study in 1998–2001 of 12 artificial reefs in one sandy area off Miami Beach, Florida, North Atlantic Ocean, USA (3) found that artificial reefs constructed with either of one of three types of material had similar abundance of spiny lobsters *Panulirus argus*. Total spiny lobster abundance was similar at reefs made of limestone boulders (16), gravel-concrete aggregate (16), and tyre-concrete aggregate (14). Between June and August 1998, twelve artificial reefs constructed with either limestone boulders, gravel-concrete aggregate, or tyre-concrete aggregate (four of each) were created at 7 m water depth. Every two months between October 1998 and February 2001, one diver recorded the total abundance of spiny lobster at each reef.

A replicated, controlled study in 2005–2006 of six sites with artificial reef modules in the Firth (Lynn) of Lorn, west coast of Scotland, UK (4) found that the complexity of the modules had mixed effects on the abundances of edible crab *Cancer pagurus* and velvet swimming crab *Necora puber*, which varied with the seasons. For edible crabs, abundances in summer and autumn were similar at all modules (0–0.05/m^2). In winter, abundance was higher at complex modules (0.13/m^2), than simple modules (0.04/m^2). In spring, abundance at complex modules (0.15/m^2) was not significantly higher than at simple modules (0.04/m^2). For swimming crabs, abundance in summer was similar at both module types (0.15–0.27/m^2). In all other seasons, abundance was higher at complex modules (0.34–0.45/m^2), than simple modules (0.05–0.18/m^2). In 2003–2004, an artificial reef complex made of multiple modules of either simple solid concrete blocks or complex perforated blocks was created. Six sites were surveyed: three simple modules and three complex modules. Monthly in August 2005–June 2006, divers recorded edible and swimming crab abundances along two 9 m^2 belt transect/site. Data were grouped by season. A prior study showed that habitat complexity was higher on complex modules than simple modules.

A controlled study in 2005–2008 of an artificial reef complex made of pyramids and plinth-poles created on soft seabed 3 nm off the coast of Italy, Mediterranean Sea (5) found that during the three years following creation, "sea-friendly" concrete pyramids developed a significantly different invertebrate community composition compared to traditional concrete plinth-pole structures. Invertebrate community composition remained dissimilar between the two structure types over the three years (year 1: 40% similarity; year 2: 73%; year 3: 68%). For the first two years, pyramids had lower invertebrate species richness (average 10 species) and abundance (average 2 individuals/dm^2) compared to plinth-pole structures (richness: 27; abundance: 48). After three years, pyramids had similar species richness (27) and higher abundance (85) compared to plinth-pole structures (richness: 24; abundance: 33). Diversity (reported as a diversity index) was lower on pyramids after a year compared to plinth-pole structures, higher after two, and not different after three. In 2005, the Pedaso artificial reef, made of 76 pyramids of "sea-friendly" concrete slabs surrounded by 214 plinth-pole structures made of traditional concrete (aimed at preventing illegal trawling), was created at 15 m depth (see paper for details on reef architecture). Invertebrates colonizing the reef were surveyed in summer in 2006, 2007 and 2008 (three surveys/year). During each survey, divers scraped a 40 × 40 cm area on the external vertical sides of three randomly-chosen structures for each reef type, and invertebrates (>0.5 mm) were identified, counted and weighed.

A replicated, controlled study in 2006–2012 of an unspecified number of artificial reefs made of pyramids and tubes created on muddy seabed 2 nm offshore of the Po River Delta, northern Mediterranean Sea, Italy (6) found that "sea-friendly" concrete pyramid reefs developed a different invertebrate community composition compared to bundle reefs of traditional concrete tubes, after 2–6 years depending on creation year. For reefs deployed in 2006, pyramid reefs had similar communities to bundle reefs by 2009, but different ones by 2012. For reefs deployed in 2010, pyramid reefs developed different communities to bundle reefs by 2012. Community data were presented as graphical analyses. In addition, at all times, species richness was higher on bundle reefs (38–55 species/sample) compared to pyramid reefs (33–45 species/sample). In 2006 and 2010, artificial reefs made of either pyramids of "sea-friendly" concrete slabs or bundles of traditional concrete tubes were created at 13–14 m depths (see paper for details). Divers surveyed the external sides of four randomly-chosen structures for each reef type in 2009 and 2012. For each structure, invertebrates were identified and counted from four 20 × 20 cm quadrats. They were also identified, and their percent cover estimated from six photographs/structure (21 × 26 cm).

A replicated, controlled study in summer 2000 and 2004 of one artificial reef made of modules off southern California, North Pacific Ocean, USA (7) found that from one to five years after their deployment, there was no difference in species richness or abundance (as % cover) of invertebrates growing on modules made of granite and those made of concrete (results presented as statistical model output). The artificial reef was created to compensate for the loss of giant kelp forest in California. Low lying (<1 m tall) artificial reef modules (40 × 40 m) made of either granite boulders of concrete rubble boulders were put on the seabed in seven sites in 1999 (4 modules/material/site) at 13–16 m depth. Invertebrate communities were sampled after one and five years. Invertebrate abundance was assessed for 42 of the 56 modules using six 1 m^2 quadrats/modules.

A replicated, controlled study in 2008–2010 of four artificial breakwaters in northwest Mobile Bay, Gulf of Mexico, Alabama, USA (8) found that breakwaters made of bags of oyster shells recruited more oysters and ribbed mussels, but did not have different species richness and abundance of small mobile animal species (invertebrates and fish combined, referred to as "nekton"), compared to "ReefBall" breakwaters, during the two years following deployment. More eastern oysters *Crassostrea virginica* were recorded on shell breakwaters (20 in total) than on ReefBall breakwaters (2) throughout the study period (data not statistically tested). On average across the study period, significantly more ribbed mussels *Geukensia demissa* were recorded on shell breakwaters (>2,500/m^2) than on ReefBall breakwaters (14/m^2). Across the study period, shell and ReefBall breakwaters had similar nekton species richness (shell: 2.3; ReefBall: 2.2 species/m^2) and abundance (shell: 0.46; ReefBall: 0.46 individual/m^2). Four artificial breakwaters made of either bags of clean oyster shells (2,000 bags/breakwater) or ReefBall modules (three rows of 41 modules/breakwater), acting as artificial reefs, were created in May 2008 along an eroding shoreline in Mobile Bay (60 m from, and parallel to, the shore; 0.75m depth). On three occasions in 2008–2010, nine modules/ReefBall breakwater and nine bags/shell breakwater were sampled. The surface area of each modules and the content of each bag were examined for live oysters and mussels. Between May 2008 and November 2009, nekton was surveyed on each side of all breakwaters using a bag seine (6.25 mm mesh) deployed over 12.5 m. All organisms were identified and counted.

(1) Collins K.J., Jensen A.C., Mallinson J.J., Roenelle V. & Smith I.P. (2002) Environmental impact assessment of a scrap tyre artificial reef. *ICES Journal of Marine Science*, 59, 243–249.
(2) Kress N., Tom M. & Spanier E. (2002) The use of coal fly ash in concrete for marine artificial reefs in the southeastern Mediterranean: compressive strength, sessile biota, and chemical composition. *ICES Journal of Marine Science*, 59, 231–237.

(3) Walker B.K., Henderson B. & Spieler R.E. (2002) Fish assemblages associated with artificial reefs of concrete aggregates or quarry stone offshore Miami Beach, Florida, USA. *Aquatic Living Resources*, 15, 95–105.
(4) Hunter W.R. & Sayer M.D.J. (2009) The comparative effects of habitat complexity on faunal assemblages of northern temperate artificial and natural reefs. *ICES Journal of Marine Science*, 66, 691–698.
(5) Spagnolo A., Cuicchi C., Punzo E., Santelli A., Scarcella G. & Fabi G. (2014) Patterns of colonization and succession of benthic assemblages in two artificial substrates. *Journal of Sea Research*, 88, 78–86.
(6) Ponti M., Fava F., Perlini R.A., Giovanardi O. & Abbiati M. (2015) Benthic assemblages on artificial reefs in the northwestern Adriatic Sea: Does structure type and age matter? *Marine Environmental Research*, 104, 10–19.
(7) Schroeter S.C., Reed D.C. & Raimondi P.T. (2015) Effects of reef physical structure on development of benthic reef community: A large-scale artificial reef experiment. *Marine Ecology Progress Series,* 540, 43–55.
(8) Scyphers S.B., Powers S.P. & Heck K.L. (2015) Ecological value of submerged breakwaters for habitat enhancement on a residential scale. *Environmental Management*, 55, 383–391.

12.12. Locate artificial reefs near aquaculture systems to benefit from nutrient run-offs

- **Two studies** examined the effects of locating artificial reefs near aquaculture systems to benefit from nutrient run-offs on subtidal benthic invertebrate populations. One study was in the Gulf of Aqaba[1] (Israel and Jordan), and one in the Mediterranean Sea[2] (Spain).

COMMUNITY RESPONSE (1 STUDY)

- **Overall community composition (1 study):** One controlled study in the Mediterranean Sea[2] found that an artificial reef located under aquaculture cages had similar invertebrate community composition to artificial reefs located at sites without aquaculture cages.

POPULATION RESPONSE (1 STUDY)

- **Overall abundance (1 study):** One controlled study in the Gulf of Aqaba[1] found that an artificial reef located at an aquaculture site had similar invertebrate biomass growing on it compared to an artificial reef located at a site without aquaculture cages.

Background

Artificial reefs are man-made structures intentionally put into the marine environment to act similarly to a natural reef. Originally used to improve fisheries and biological resources, they have been shown to be ecologically beneficial by locally enhancing biodiversity and increasing abundance (Bohnsack & Sutherland 1985; Clark & Edwards 1999). By locating an artificial reef near an aquaculture system, the introduced reef can potentially benefit from the nutrient-rich run-offs frequently associated with aquaculture systems, thereby promoting the colonisation and development of a subtidal benthic invertebrate community on the reef (Aguado-Giménez *et al*. 2011).

Evidence for related interventions is summarised under “Threat: Pollution – Locate artificial reefs near aquaculture systems to act as biofilters” and “Threat: Energy production and mining – Co-locate aquaculture systems with other activities and infrastructures (such as windfarms)”.

Aguado-Giménez F., Piedecausa M.A., Carrasco C., Gutiérrez J.M., Aliaga V. & García-García B. (2011) Do benthic biofilters contribute to sustainability and restoration of the benthic environment impacted by offshore cage finfish aquaculture? *Marine Pollution Bulletin*, 62, 1714–1724.
Bohnsack J.A. & Sutherland D.L. (1985) Artificial reef research: a review with recommendations for future priorities. *Bulletin of Marine Science*, 37, 11–39.

Clark S. & Edwards A.J. (1999) An evaluation of artificial reef structures as tools for marine habitat rehabilitation in the Maldives. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 9, 5–21.

A controlled study in 1999–2000 of two artificial reefs in an area of sand and seagrass in the Gulf of Aqaba (Gulf of Eilat), Red Sea, Israel and Jordan (1) found that, after one year, an artificial reef deployed at an aquaculture site did not appear to develop a higher biomass of sessile invertebrates compared to an artificial reef deployed at a site without aquaculture activity. Data were not statistically tested. Biomass of invertebrates varied between the two reefs following deployment and tended to be similar after a year (approximately 700 kg/reef). In March 1999, two artificial reefs (8.2 m^3) were deployed at 20 m depth: one at a fish farm, and another at a site 500 m west of the fish farm without aquaculture activity. Each reef held multiple 30 × 45 cm sample plates used for invertebrates to colonise. Three plates were sampled monthly from each artificial reef, photographed, dried, and the biomass of attached invertebrates recorded.

A controlled study in 2006–2007 of three sites in one soft seabed area off the coast of Murcia, Mediterranean Sea, southeastern Spain (2) found that, after one year, an artificial reef deployed underneath aquaculture cages did not develop a more diverse invertebrate community compared to artificial reefs deployed at sites without aquaculture cages. Invertebrate community composition varied during the year following deployment, but the artificial reef located under the cages had similar invertebrate community composition to those located away from the cages at each sampling time (data presented as statistical model results). In May 2006, three biofilter-like artificial reefs were deployed at 37–38 m depths: one underneath aquaculture cages, and two at sites without cages located 1.3 and 1 km away from the aquaculture site respectively. Each reef held multiple 30 × 30 cm sample units. Four randomly-chosen units were sampled by divers in summer and autumn 2006, winter 2006/07, and spring and summer 2007, at each reef. Invertebrates growing on the structures were identified and counted for each unit.

(1) Angel D.L., Eden N., Breitstein S., Yurman A., Katz T. & Spanier E. (2002) In situ biofiltration: a means to limit the dispersal of effluents from marine finfish cage aquaculture. *Hydrobiologia*, 469, 1–10.
(2) Aguado-Giménez F., Piedecausa M.A., Carrasco C., Gutiérrez J.M., Aliaga V. & García-García B. (2011) Do benthic biofilters contribute to sustainability and restoration of the benthic environment impacted by offshore cage finfish aquaculture? *Marine Pollution Bulletin*, 62, 1714–1724.

12.13. Place anthropogenic installations (e.g: windfarms) in an area such that they create artificial habitat and reduce the level of fishing activity

- We found no studies that evaluated the effects of placing anthropogenic installations in an area such that they reduce the level of fishing activity on subtidal benthic invertebrate 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

Anthropogenic installations, such as wind farms, drilling platforms, rigs, can act as artificial reefs, by introducing hard and complex habitats often where the natural habitat is soft sediment (Krone *et al*. 2017; Langhamer & Wilhelmsson 2009). These installations may have safety zones around them, either temporarily or permanently, to limit, reduce or remove human activities that can take place, such as fishing. While this may be due to the risk of gear loss and snagging when in contact with such installations, it can have secondary conservation benefits by acting as *de facto* marine reserves (Ashley *et al.* 2014). By both acting as artificial reefs and reducing the level of fishing activity around them, placing anthropogenic installations in an area can reduce threats to subtidal benthic invertebrates while promoting their recovery.

Related evidence for the effects of creating buffer zones around anthropogenic installations is summarised under "Threat: Transportation and service corridors – Cease or prohibit shipping".

Evidence for the co-location of activities and marine spatial planning is summarised under "Threat: Energy production and mining – Co-locate aquaculture systems with other activities and infrastructures (such as wind farms) to maximise use of marine space", "Threat: Pollution – Locate artificial reefs near aquaculture systems to act as biofilters", and "Habitat restoration and creation – Locate artificial reefs near aquaculture systems to benefit from nutrient run-offs". Further evidence related to creating artificial reefs is summarised under "Habitat restoration and creation – Artificial habitat creation", while evidence related to regulating and mitigation fishing is summarised under "Threat: Biological resource use".

Ashley M.C., Mangi S.C. & Rodwell L.D. (2014) The potential of offshore windfarms to act as marine protected areas–a systematic review of current evidence. *Marine Policy*, 45, 301–309.

Krone R., Dederer G., Kanstinger P., Krämer P., Schneider C. & Schmalenbach I. (2017) Mobile demersal megafauna at common offshore wind turbine foundations in the German Bight (North Sea) two years after deployment-increased production rate of Cancer pagurus. *Marine Environmental Research*, 123, 53–61.

Langhamer O. & Wilhelmsson D. (2009) Colonisation of fish and crabs of wave energy foundations and the effects of manufactured holes–a field experiment. *Marine Environmental Research*, 68, 151–157.

12.14. Repurpose obsolete offshore structures to act as artificial reefs

- **One study** examined the effects of repurposing obsolete offshore structures on subtidal benthic invertebrates. The study was of a sunken oil rig in the Mediterranean Sea[1] (Italy).

COMMUNITY RESPONSE (1 STUDY)

- **Overall species richness/diversity (1 study):** One study in the Mediterranean Sea[1] recorded at least 53 invertebrate species having colonised a sunken oil rig after 30 years. Species included 14 species of molluscs, 14 species of worms, and 11 species of crustaceans.

POPULATION RESPONSE (0 STUDIES)

Background

Offshore structures (such as oil rigs and windfarm turbine bases), have been shown to support diverse invertebrate assemblages, and act as artificial reefs (Coates *et al*. 2014; Krone *et al*. 2017; Langhamer & Wilhelmsson 2009). For instance, wind turbine foundations in a windfarm in the German Bight, North Sea, were found to host over 5,000 edible crabs *Cancer pagurus* per foundation (Krone *et al*. 2017).

Following decommissioning, entire or parts of obsolete offshore structures can potentially be repurposed to act as artificial reefs instead of being removed from the marine environment (Frumkes 2002). By repurposing obsolete offshore structures within the marine environment and leaving them in place or placing them in strategic locations, they can potentially provide additional hard surface for subtidal benthic invertebrates to colonise and seek shelter, and thereby enhance local invertebrate biodiversity (Frumkes 2002; Ponti *et al.* 2002). This can also benefit subtidal benthic invertebrates by avoiding or reducing the disturbance associated with the removal of these structures from the marine environment.

Evidence for interventions relating to the decommissioning of offshore structures is summarised under "Threat: Energy production and mining – Leave pipelines and infrastructure in place following decommissioning" and "Threat: Transportation and service corridors – Leave utility and service lines in place after decommissioning". Evidence related to artificial reefs is summarised under "Habitat restoration and creation – Habitat enhancement" and "Habitat restoration and creation – Artificial habitat creation".

Coates D.A., Deschutter Y., Vincx M. & Vanaverbeke J. (2014) Enrichment and shifts in macrobenthic assemblages in an offshore wind farm area in the Belgian part of the North Sea. *Marine Environmental Research*, 95, 1–12.

Frumkes D.R. (2002) The status of the California Rigs-to-Reefs Programme and the need to limit consumptive fishing activities. *ICES Journal of Marine Science*, 59, 272–276.

Krone R., Dederer G., Kanstinger P., Krämer P., Schneider C. & Schmalenbach I. (2017) Mobile demersal megafauna at common offshore wind turbine foundations in the German Bight (North Sea) two years after deployment-increased production rate of Cancer pagurus. *Marine Environmental Research*, 123, 53–61.

Langhamer O. & Wilhelmsson D. (2009) Colonisation of fish and crabs of wave energy foundations and the effects of manufactured holes–a field experiment. *Marine Environmental Research*, 68, 151–157.

Ponti M., Abbiati M. & Ceccherelli V.U. (2002) Drilling platforms as artificial reefs: distribution of macrobenthic assemblages of the ''Paguro'' wreck (northern Adriatic Sea). *ICES Journal of Marine Science*, 59, 316–323.

A study in 1994 in the northern Mediterranean Sea, Italy (1) found that a drilling platform that had been left in place to act as an artificial reef after sinking was colonised by at least 53 invertebrate species. Species included 14 species of molluscs, 14 species of worms, and 11 species of crustaceans. Most recorded species were associated with the hard habitat created by mussels and oysters. The drilling platform sank in 1965 due to a fire. The area surrounding it was then declared a marine protected area prohibiting all fishing. Samples were collected in summer 1994 between 10 and 34 m depths using two methods. Divers manually scraped off three 20 x 20 cm areas from each of four sites (two orientations within two water depths). Invertebrates (>0.5 mm) were identified and counted. Divers also took photographs along five vertical transects. Percentage cover of organisms were estimated from the photographs.

(1) Ponti M., Abbiati M. & Ceccherelli V.U. (2002) Drilling platforms as artificial reefs: distribution of macrobenthic assemblages of the ''Paguro'' wreck (northern Adriatic Sea). *ICES Journal of Marine Science*, 59, 316–323.

Other habitat restoration and creation interventions

12.15. Pay monetary compensation for habitat damage remediation

- We found no studies that evaluated the effects of paying monetary compensation for habitat damage remediation on subtidal benthic invertebrate 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

Following damages to habitats and natural resources, for instance from pollution or physical disturbances, the guilty parties could be required to pay monetary compensation (Price *et al*. 2012). Such compensations can be direct monetary 'fines' for damages, payments towards the costs of environmental assessments, or reimbursement for remediation, restoration or offsetting actions. For instance, following the massive oil spill of 1991, a multibillion-dollar compensation was paid to several Gulf States for damages and restoration projects (Payne & Sand 2011; Price *et al.* 2012). Monetary compensation can benefit subtidal benthic invertebrates when used for remediation, restoration or offsetting projects (Jones *et al.* 2012).

Evidence related to offsetting projects for lost habitats, which can be undertaken through monetary compensation, are summarised under "Habitat restoration and creation – Offset habitat loss from human activity by restoring or creating habitats elsewhere".

Jones D.A., Nithyanandan M. & Williams I. (2012) Sabah Al-Ahmad Sea City Kuwait: development of a sustainable manmade coastal ecsystem in a saline desert. *Aquatic Ecosystem Health Management*, 15, 82–90.

Payne C. & Sand P. (Eds.) (2011) *Gulf War Reparations and the UN Compensation Commission Environmental Liability*. Oxford University Press, Oxford.

Price A.R.G., Donlan M.C., Sheppard C.R.C. & Munawar M. (2012) Environmental rejuvenation of the Gulf by compensation and restoration. *Aquatic Ecosystem Health & Management*, 15, 7–13.

12.16. Remove and relocate habitat-forming (biogenic) species before onset of impactful activities

- **One study** examined the effects of removing and relocating habitat-forming species before onset of impactful activities on subtidal benthic invertebrates. The study was in the Fal Estuary[1] (UK).

COMMUNITY RESPONSE (1 STUDY)

- **Overall community composition (1 study):** One replicated, paired, controlled study in the Fal Estuary[1] found that invertebrate community composition was different in plots where maërl bed habitat had been removed and relayed compared to undisturbed maërl after five weeks, but similar after 44 weeks.
- **Overall species richness/diversity (1 study):** One replicated, paired, controlled study in the Fal Estuary[1] found that invertebrate species richness was lower in plots where maërl bed habitat had been removed and relayed compared to undisturbed maërl after five weeks, but similar after 44 weeks.

POPULATION RESPONSE (1 STUDY)

- **Overall abundance (1 study):** One replicated, paired, controlled study in the Fal Estuary[1] found that invertebrate abundance was different in plots where maërl bed habitat had been removed and relayed compared to undisturbed maërl after five weeks, but similar after 44 weeks.

Background

Marine biogenic habitats are habitats created by the occurrence of specific marine species, such as coral reefs, oyster reefs, mussel beds, or kelp forests (Jones *et al.* 1994). They form a new complex environment for other species to live in and can locally promote subtidal benthic invertebrate biodiversity. Many populations of marine biogenic species have declined or been depleted due to the multiple threats they are under, including habitat damage or loss and direct physical damages from anthropogenic activities (Airoldi *et al*. 2009). As a pre-emptive conservation measure, biogenic species can potentially be temporarily removed to allow for an impactful activity to occur, then relocated back into their original location, or at a different location. Such measures have been trialled to preserve maërl habitat and its associated faunal community during the dredging of new shipping channels (Sheehan *et al*. 2015).

When this intervention is undertaken for species which do not form habitats, evidence has been summarised under "Species management – Remove and relocate invertebrate species before onset of impactful activities".

Airoldi L., Connell S.D. & Beck M.W. (2009) The loss of natural habitats and the addition of artificial substrata. Pages 269–280 in: Wahl M. (eds) *Marine Hard Bottom Communities*. Springer, Berlin, Heidelberg.

Jones C.G., Lawton J.H. & Shachak M. (1994) Organisms as ecosystem engineers. Pages 130–147 in: *Ecosystem Management*. Springer, New York, NY.

Sheehan E.V., Bridger D., Cousens S.L. & Attrill M.J. (2015) Testing the resilience of dead maerl infaunal assemblages to the experimental removal and re-lay of habitat. *Marine Ecology Progress Series*, 535, 117–128.

A replicated, paired, controlled study in 2012–2013 of 24 plots in six sites of maërl bed in the Fal Estuary, southwest England, UK (1) found that plots where maërl bed habitat had been temporarily removed then relayed had fewer invertebrate species, reduced abundance, and a different community composition, compared to plots of undisturbed maërl, after five weeks but not after 44 weeks. After five weeks, the removed-relayed plots had fewer species (54 species/core) and lower abundance (155 individuals/core) compared to undisturbed maërl plots (species: 94; abundance: 282), and a different community composition (community data presented at statistical model results and graphical analyses). After 44 weeks, species richness and abundance were similar in the removed-relayed plots (species: 93; abundance: 263) and the undisturbed maërl plots (species: 91; abundance: 178), and community compositions were similar. Dredging of shipping lanes was planned in Falmouth Harbour. This trial study aimed to assess the feasibility of removing and relaying maërl as a mitigation action prior to dredging. Four 5 m^2 plots were selected at each of six sites. One of two treatments was attributed to each plot: maërl removed then relayed, undisturbed maërl (representing natural maërl where no dredging for shipping lane occurred). In September 2012, the top 0.3 m of maërl was dredged from the removed-relayed plots and relayed to its original position 12 h later (to mimic the duration of shipping lane dredging). Five maërl samples were collected using a hand corer (25 cm length, 10 cm diameter) from one plot/treatment/site after five and 44 weeks. Invertebrates associated with maërl habitat (>0.5 mm) were counted.

(1) Sheehan E.V., Bridger D., Cousens S.L. & Attrill M.J. (2015) Testing the resilience of dead maerl infaunal assemblages to the experimental removal and re-lay of habitat. *Marine Ecology Progress Series*, 535, 117–128.

12.17. Offset habitat loss from human activity by restoring or creating habitats elsewhere

- **Two studies** examined the effects of offsetting habitat loss from human activity by restoring or creating habitats elsewhere on subtidal benthic invertebrate populations. One study was in the Delaware Bay[1] (USA), the other in the Persian Gulf[2] (Kuwait).

COMMUNITY RESPONSE (1 STUDY)

- **Overall richness/diversity (1 study):** One study in the Persian Gulf[2] found that an area of low ecological value restored to offset habitat lost to land reclamation was colonized by over 198 invertebrate species.

POPULATION RESPONSE (0 STUDIES)

OTHER (1 STUDY)

- **Biological production (1 study):** One study in Delaware Bay[1] found that an artificial reef built to offset lost soft-sediment habitat had higher annual secondary production/unit area from sessile invertebrates, but lower total annual secondary production, compared to habitat similar to that lost.

Background

Habitat restoration and creation interventions can be undertaken at a site as a biodiversity offset strategy to replace the biodiversity lost at an impacted site, with the aim to achieve 'no net loss' of overall biodiversity (Ives & Bekessy 2015). Offsetting can be done "in-kind", where the new habitat is similar to the lost one (for example creating coral reefs to replace lost corals reef elsewhere), or "out-of-kind", where the new habitat is different to the lost one (for example creating an artificial reef to replace lost mudflats elsewhere; evidence also summarised under "Habitat restoration and creation – Create artificial reefs" and "Create artificial reefs of different 3-D structure and material used"). When offsetting is undertaken in the marine and coastal environment, it can help promote subtidal benthic invertebrate biodiversity (Jones *et al.* 2007).

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

Jones D.A., Ealey T., Baca B., Livesey S. & Al-Jamali F. (2007) Gulf desert developments encompassing a marine environment, a compensatory solution to the loss of coastal habitats by infill and reclamation: The case of the Pearl City Al-Khiran, Kuwait. *Aquatic Ecosystem Health & Management*, 10, 268–276.

A study in 1990–1994 of an offset site in Delaware Bay, USA (1) found that a 0.7 ha artificial reef created to offset 57.4 ha of lost soft-sediment habitat had higher annual secondary production/unit area from sessile invertebrates, but lower total annual secondary production, compared to habitat similar to that lost. Annual secondary production/unit area at the offset artificial reef was 11–67 times higher than at soft-sediment habitat similar to that lost (2,000–12,000 kcal/m^2/year vs 177 kcal/m^2/year), but total annual secondary production was 1.3–7.6 times lower (13–77 million cal/year vs 100 million cal/year). The authors concluded that the artificial reef improved secondary production, but not enough reef was created to fully offset the lost habitat. An artificial reef was created in 1989 to mitigate the loss of mudflats elsewhere following the creation of a dredged material disposal site. Twice per summer in 1990–1994, sessile invertebrates growing on the artificial reef were identified and their biomass measured (see original paper for methods). As the mudflat habitat had already been lost, data for comparison were collected from similar soft-sediment habitats in 1990–1993 by the US

Protection Agency and the Chesapeake Bay Program. For each habitat type, biomass data were used to estimate annual secondary production.

A study in 2004–2005 of three sites in one area of soft seabed in the Persian Gulf, Kuwait (2) found that restoring a 30 km^2 area of low ecological value to offset 20 km^2 of habitat lost to land reclamation led to the creation of 1.27 km^2 of subtidal channels that were colonized by over 198 benthic invertebrate species within a year. Al-Khiran Pearl Sea City was constructed following a biodiversity offsetting approach by creating waterways, beaches and planted areas (mangrove, seagrass, saltmarsh). One waterway was opened to the sea in 2004. In 2005, invertebrates at three sites within the waterway were surveyed using a variety of methods (see original paper) including a 0.15 m^3 sediment grab and a 50 m^2 dredge. Invertebrates (>0.5 mm) were identified.

(1) Burton, W.H., Farrar, J.S., Steimle, F., & Conlin, B. (2002) Assessment of out-of-kind mitigation success of an artificial reef deployed in Delaware Bay, USA. *ICES Journal of Marine Science*, 59, 106–110.
(2) Jones D.A., Ealey T., Baca B., Livesey S. & Al-Jamali F. (2007) Gulf desert developments encompassing a marine environment, a compensatory solution to the loss of coastal habitats by infill and reclamation: The case of the Pearl City Al-Khiran, Kuwait. *Aquatic Ecosystem Health & Management*, 10, 268–276.

13. Species management

Background

This chapter describes interventions that can be used to increase the population size of specific marine subtidal benthic invertebrate species, for instance by translocating wild individuals from one area to another, breeding or rearing individuals in captivity (*ex-situ* conservation) then releasing them into the wild, or by instating catch quotas if a species is recreationally harvested (Caddy *et al.* 2003). Interventions related to harvest restrictions or to the management of specific species within a Marine Protected Area are described in "Habitat protection". Please note that specific interventions aimed to promote the populations of commercially targeted species, and as such more closely related to harvest and fisheries management, are not included in this synopsis. Therefore, the outcome of interventions such as "Set recreational/commercial catch quotas" for the commercial species are not summarised here. We make one exception when the intervention is to stop the fishery altogether, for instance "ceasing or prohibiting harvest of conch" to conserve the conch population. Note as well that in the case of habitat-forming (biogenic) subtidal benthic invertebrate species, such as oysters or honeycomb worms (Airoldi *et al*. 2008; Jones *et al.* 1994), interventions that aim to increase associated invertebrate biodiversity (by recreating or restoring the habitats they form) are described in "Habitat restoration and creation", while interventions that aim to increase the habitat-forming species (meaning, the oysters or the worms themselves) are described here.

Airoldi L., Balata D. & Beck M.W. (2008) The gray zone: relationships between habitat loss and marine diversity and their applications in conservation. *Journal of Experimental Marine Biology and Ecology*, 366, 8–15.

Caddy J.F., Defeo D. & Defeo O. (2003) *Enhancing or Restoring the Productivity of Natural Populations of Shellfish and Other Marine Invertebrate Resources* (Vol. 448). Food & Agriculture Organisation.

Jones, C.G., Lawton, J.H., & Shachak, M. (1994) Organisms as ecosystem engineers. Pages 130–147 in: *Ecosystem Management*. Springer, New York, NY.

13.1. Transplant/release captive-bred or hatchery-reared species

Background

Many populations of subtidal benthic invertebrate species have declined or been depleted due to the multiple threats they are under, such as habitat loss and overharvest (Airoldi *et al*. 2008; Hobday *et al*. 2000). To counteract this phenomenon, captive-bred or hatchery-reared subtidal benthic invertebrates can be transplanted or released at a site, either to introduce a species to a new site (where they did not historically occur), to reintroduce a species to a site (where they used to occur), or to enhance the population at a site where the species is already present by increasing its abundance (Hansen & Gosselin 2013). As the outcomes of transplanting/releasing species can vary largely with the type of species, studies have been grouped by broader taxonomic group (e.g: crustaceans such as lobsters or prawns; or molluscs such as abalone, scallops, or mussels).

Here, only direct transplanting/releasing methods have been considered, without added interventions or changes to the methods to improve survival after release.

However, when transplant/release was undertaken in predator exclusion cages, evidence has been summarised under "Species management – Transplant/release captive-bred or hatchery-reared species in predator exclusion cages".

When transplanting hatchery-reared individuals is undertaken for a habitat-forming species, effects on the invertebrates associated with the habitat are reported in "Habitat restoration and creation – Transplant captive-bred or hatchery-reared habitat-forming (biogenic) species". Evidence from translocation studies of naturally occurring species is summarised under "Species management – Translocate species" and "Habitat restoration and creation – Translocate habitat-forming (biogenic) species".

Airoldi L., Balata D. & Beck M.W. (2008) The gray zone: relationships between habitat loss and marine diversity and their applications in conservation. *Journal of Experimental Marine Biology and Ecology*, 366, 8–15.

Hansen S.C. & Gosselin L.A. (2013) Do predators, handling stress or field acclimation periods influence the survivorship of hatchery-reared abalone *Haliotis kamtschatkana* outplanted into natural habitats? *Aquatic Conservation: Marine and Freshwater Ecosystems,* 23, 246–253.

Hobday A.J., Tegner M.J. & Haaker P.L. (2000) Over-exploitation of a broadcast spawning marine invertebrate: decline of the white abalone. *Reviews in Fish Biology and Fisheries*, 10, 493–514.

13.1.1. Transplant/release crustaceans

- **Five studies** examined the effects of transplanting or releasing hatchery-reared crustacean species on their wild populations. Four examined lobsters in the North Sea[1-4] (Germany, Norway, UK), and one examined prawns in the Swan-Canning Estuary[5] (Australia).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (5 STUDIES)

- **Crustacean abundance (1 study):** One study in the Swan-Canning Estuary[5] found that after releasing hatchery-reared prawn larvae into the wild, the abundance of egg-bearing female prawns increased.
- **Crustacean reproductive success (3 studies):** Two studies (one controlled) in the North Sea[3,4] found that after their release, recaptured hatchery-reared female lobsters carried eggs[3,4], and the number, size and developmental stage of eggs were similar to that of wild females[3]. One study in the Swan-Canning Estuary[5] found that after releasing hatchery-reared prawn larvae into the wild the overall population fecundity (egg production/area) increased.
- **Crustacean survival (2 studies):** Two studies in the North Sea[1,4] found that 50–84%[1] and 32–39%[4] of hatchery-reared lobsters survived in the wild after release, up to eight and up to five years, respectively.
- **Crustacean condition (4 studies):** Two studies in the North Sea[1,4] found that hatchery-reared lobsters grew in the wild after release. One controlled study in the North Sea[2] found that after release into the wild, hatchery-reared female lobsters had similar growth rates as wild females. One study in the North Sea[4] found that after releasing hatchery-reared lobsters, no recaptured lobsters displayed signs of "Black Spot" disease, and 95% had developed a crusher-claw. One study in the Swan-Canning Estuary[5] found that after releasing hatchery-reared prawn larvae into the wild, the size of egg-baring female prawns increased.

BEHAVIOUR (1 STUDY)

- **Crustacean movement (1 study):** One controlled study in the North Sea[2] found that after release into the wild, hatchery-reared female lobsters had similar movement patterns as wild females.

A study in 1983–1992 in one seabed area off the east coast of England, North Sea, UK (1) estimated that between 50–84% of the initial number of released hatchery-reared European lobsters *Homarus gammarus* survived and increased in size for up to eight years in the wild. Lobsters recaught reached 85 mm (legal catch size) within four to eight years after release. Between 1983 and 1988, hatchery-reared lobsters (49,000 in total) were tagged and released across an area of 30 x 8 km onto cobbles and boulders at 80 locations (10–15 m depth). At time of release, lobsters were three months old with carapaces measuring 15 mm in length. Between 1988 and 1992, a recapture programme caught a total of 56,700 lobsters, of which 621 were tagged lobsters previously released. The carapaces of recaptured tagged lobsters were measured. It is not known if the number of uncaught tagged lobsters was due to mortality or recapture effort. Percentage survival of the 49,000 released lobsters was estimated from the recapture programme catch-rate data.

A controlled study in 1998–2000 in one area off southwestern Norway, North Sea (2 – same experimental set-up as 3) found that hatchery-reared female European lobsters *Homarus gammarus* released into the wild had similar growth rate and movement patterns, compared to wild females. Regardless of carapace length, the growth rate of hatchery-reared females (7–10 mm between moults) was similar to that of wild females (3–8 mm). In total, 53% of hatchery-reared females remained within 500 m of their release sites, which was similar to wild females (41%). Between 1990 and 1994, hatchery-reared juvenile lobsters (approximately 128,000) were released as part of a restocking program. During the fishing season each year from 1998 to 2000, egg-bearing female lobsters caught by fishers were measured (total length, carapace length), weighed, and hatchery-reared females were differentiated from wild females by the presence of tags. All females were then retagged, kept in holding pens in the sea, and released after the end of the fishing season to potentially be recaptured by fishers the following fishing season (mark-recapture). A total of 81 hatchery-reared females and 231 wild females were recaptured at least once. Locations of release and recapture sites were recorded.

A controlled study in 1996–1997 in one seabed area off southwestern Norway, North Sea (3 – same experimental set-up as 2) found that when comparing individuals of similar sizes, female hatchery-reared European lobsters *Homarus gammarus* released into the wild carried similar numbers of eggs and their eggs were of similar weight, diameter, and developmental stage, compared to wild lobsters. For further details of results see graphs in paper. Between 1990 and 1994, hatchery-reared juvenile lobsters (approximately 128,000) were released as part of a restocking program. During autumn 1996, and spring and autumn 1997, egg-bearing female lobsters were collected from commercial landings. Hatchery-reared females (104 individuals) were differentiated from wild females (111 individuals) by the presence of tags. All female lobsters were measured (carapace length), and the weight of their egg mass recorded. For each female, egg count and size were assessed from subsamples. A note was made of any developing embryos.

A study in 2000–2009 in one area of rocky seabed off Helgoland, German Bight, North Sea (4) found that after releasing one-year-old hatchery-reared European lobsters *Homarus gammarus*, they grew and survived in the wild, became reproductive, and appeared healthy. Recaptured lobsters had grown in the wild (females: 14.5–19.8; males: 16.8–21.8 mm/year) and reached 85 mm (legal catch size) within four to seven years after release. Survival rate of lobsters released in 2000 and 2001 was estimated at 32 and

39% respectively after up to five years. In addition, no recaptured lobsters displayed signs of "Black Spot" disease, 95% had developed a crusher-claw, and 16% of recaptured females carried eggs. Annually in 2000–2005, at two locations of 10 m water depth, tagged hatchery-reared lobsters were released at the surface (5,421 lobsters in total). Released lobsters weighed 1.5 g and had carapaces 15 mm long. Between 2000 and 2009, 488 of these were recaptured at least once, using lobster pots, traps, and divers. It is not known if the number of uncaught tagged lobsters was due to mortality, recapture effort, or migration outside the search zone. Recaptured lobsters were sexed, observed for signs of disease and presence of a crusher-claw, and their carapaces measured. Percentage survival was estimated from the mark-recapture programme data obtained between 2001 and 2005 for the 1,036 released in 2000 and 2001.

A study in 2013–2016 of 36 sites in the Swan-Canning Estuary, south-western Australia (5) found that during the three years after yearly releases of hatchery-reared western school prawn larvae *Metapenaeus dalli* the abundance and size of egg-bearing females, as well as the overall population egg production, increased. Abundance of egg-bearing females increased from 0.1–0.6/500 m^2 in 2013–2014 and 0.6–1.4 in 2014–2015 to 1.1–1.6 in 2015–2016. The carapace length of egg-bearing females increased from 17–20 mm in 2014–2015 to 23-24 mm in 2015–2016. Egg production (fecundity) increased from 16,000 egg/500 m^2 in 2013–2014, to 34,000 in 2014–2015 and 163,000 in 2015–2016. However, authors indicate that wild and hatchery-reared prawns could not be discerned, and therefore that the results cannot be solely attributable to the restocking programme. Yearly between December 2012 and March 2016, hatchery-reared juvenile prawns were released into the estuary (15,000 in 2012–2013; 635,000 in 2014–2015; 2 million in 2015–2016) as part of a restocking programme. Monthly in October 2013–March 2016, prawns were collected using a mix of hand nets (9 mm mesh; 570 m^2) and otter trawls (9 mm at the codend; 650 m^2) at 36 sites (two samples/site/month). Prawns were counted, sized, sexed, and egg-bearing females recorded.

(1) Bannister R.C.A., Addison J.T. & Lovewell S.R.J. (1994) Growth, movement, recapture rate and survival of hatchery-reared lobsters (*Homarus gammarus* (Linnaeus, 1758)) released into the wild on the English east coast. *Crustaceana*, 67, 156–172.

(2) Agnalt A.-L. Kristiansen T.S. & Jørstad K.E. (2007) Growth, reproductive cycle, and movement of berried European lobsters (*Homarus gammarus*) in a local stock off southwestern Norway. *ICES Journal of Marine Science,* 64, 288–297.

(3) Agnalt A.-L. (2008) Fecundity of the European lobster (*Homarus gammarus*) off southwestern Norway after stock enhancement: do cultured females produce as many eggs as wild females? *ICES Journal of Marine Science*, 65, 164–170.

(4) Schmalenbach I., Mehrtens F., Janke M. & Buchholz F. (2011) A mark-recapture study of hatchery-reared juvenile European lobsters, *Homarus gammarus*, released at the rocky island of Helgoland (German Bight, North Sea) from 2000 to 2009. *Fisheries Research*, 108, 22–30.

(5) Crisp J.A., Loneragan N.R., Tweedley J.R., D'Souza F.M.L. & Poh B. (2018) Environmental factors influencing the reproduction of an estuarine penaeid population and implications for management. *Fisheries Management and Ecology*, 25, 203–219.

13.1.2. Transplant/release molluscs

- **Eight studies** examined the effects of transplanting or releasing hatchery-reared mollusc species on their wild populations. One examined abalone in the North Pacific Ocean[4] (Canada), one examined clams off the Strait of Singapore[2] (Singapore), one examined oysters in the North Atlantic Ocean[7a-b] (USA), and four examined scallops in the North Atlantic Ocean[3,5,6] and Gulf of Mexico[1] (USA).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (8 STUDIES)

- **Mollusc abundance (2 studies):** One replicated, before-and-after study in the North Atlantic Ocean[5] found that after transplanting hatchery-reared scallops, abundance of juvenile scallops typically increased, but not that of adult scallops. Two replicated, randomized, controlled studies in the North Atlantic Ocean[7a,b] found that after releasing hatchery-reared oyster larvae, more spat initially settled using a direct technique compared to a traditional remote technique[7a], and equal number of spat settled on cleaned and natural oyster shells[7b].
- **Mollusc reproductive success (1 study):** One replicated, before-and-after study in the North Atlantic Ocean[6] found that after transplanting hatchery-reared scallops, larval recruitment increased across all areas studied.
- **Mollusc survival (5 studies):** One replicated study in the Strait of Singapore[2] found that, after transplantation in the field, aquarium-reared clams had a high survival rate. One replicated, controlled study in the North Atlantic Ocean[3] found that after transplanting hatchery-reared scallops, the number of transplanted scallops surviving decreased regardless of the methods used, and maximum mortalities was reported to be 0–1.5%. One replicated, controlled study in the North Pacific Ocean[4] found that transplanting hatchery-reared abalone into the wild reduced survivorship compared to non-transplanted hatchery-reared abalone kept in tanks. Two replicated, randomized, controlled studies in the North Atlantic Ocean[7a,b] found that after releasing hatchery-reared oyster larvae, 61% of the settled spat survived the winter[7a], and settled spat survived equally on cleaned and natural oyster shells[7b].
- **Mollusc condition (3 studies):** Two replicated studies in the Strait of Singapore[2] and the North Atlantic Ocean[3] found after transplantation in the wild, aquarium-reared clams[2] and hatchery-reared scallops[3] increased in weight and/or grew. Scallops grew in both free-planted plots and suspended bags but grew more in free-planted plots[3]. One replicated, before-and-after study in the Gulf of Mexico[1] found that after transplanting hatchery-reared scallops, wild populations had not become genetically more similar to hatchery-reared scallops. One replicated, controlled study in the North Atlantic Ocean[3] found that after transplanting hatchery-reared scallops, free-planted scallops developed less shell biofouling than suspended scallops.

A replicated, before-and-after study in 1997–2001 in six sites of soft seabed in west-central Florida, Gulf of Mexico, USA (1) found that one year after transplanting hatchery-reared bay scallops *Argopecten irradians* to three depleted sites, populations of wild (not transplanted) bay scallops at the transplant sites and at three adjacent sites had not become genetically more similar to hatchery-reared scallops. A year after transplant, the frequency of wild bay scallops genetically similar to hatchery-reared ones (as number of haplotypes/sample) did not significantly increase in transplant sites (before: 0–3 in samples of 35–249; after: 0–5 in samples of 63–249), or across the region (transplant sites and adjacent sites combined – before: 5–12 in samples of 160–600; after: 13–23 in samples of 512–991). Between 1998 and 2000, hatchery-reared bay scallops (23,000–63,000/site; 20–30 mm in length) were transplanted in cages (50/site) within seagrass beds to three depleted sites during three transplant events (see study for details). Divers collected wild bay scallops (50–300/site; 40 mm in length) before and a year after each transplantation events at all transplant sites and at three adjacent sites (without transplants but benefitting from spill-over effect). Scallops were genetically assessed and compared to hatchery-reared scallops.

A replicated study in 2004 at four coral reef sites in the Singapore Strait (2) found that after being transplanted in the field aquarium-reared giant clams *Tridacna squamosa* had a high survival rate and grew over seven months. Of the 144 clams transplanted, 116 were recovered (80.6%), but survival rates differed amongst transplant sites (24–34 out of 36 transplanted clams per site). All recovered clams had increased in weight, shell length and shell height over the seven-month transplant, but rates differed amongst transplant sites (3.3–4.8 mm/month). In April 2004, a total of 144 aquarium-reared clams (eighteen-month old) were equally divided into 24 groups (6 clams/group) and transplanted into four sites (6 groups/site). Clams were released 50 cm above the seabed. Prior to transplant and after seven months, all clams were weighted, and their shell lengths and heights measured.

A replicated, controlled study in 2005–2006 in one area of muddy sandy seabed with in Northwest Harbor, North Atlantic Ocean, New York, USA (3) found that over six months after transplanting hatchery-reared bay scallops *Argopecten irradians irradians*, abundance (indicating survival) decreased in plots where they were free-planted and in suspended bags, and that scallop growth and formation of shell biofouling varied with transplantation method. In both years, abundance of free-planted scallops decreased over time (2005: from 81–110/m^2 to 18–37/m^2; 2006: from 65–253/m^2 to <1/m^2). Authors report maximum mortalities of 0–1.5%. In both years, abundance of suspended scallops decreased over time (data presented on a logarithm scale), and typically did not vary with stocking densities (7 of 11 sampling dates/year; data not shown). Changes in abundances was not compared between transplanting methods. Transplanted scallops grew in both methods over 6.5 months but grew more in free-planted plots (2005: +20–21 mm; 2006: +30–33 mm) compared to suspended bags (2005: +13–14 mm; 2006: +21–22 mm). Growth rate of scallops in bags did not vary with stocking densities (data not shown). Over the 4.5 months after transplantation, free-planted scallops developed less biofouling than suspended scallops (2005: 0.62 vs 1.98 g/scallop; 2006: 0.91 vs 2.5 g/scallops; data extracted from the text). Two methods of transplantation were tested: free-planting and suspended bags. Free-planted scallops were distributed directly on the seabed in four 25 x 25 m plots at 1.3–3 m depth. Suspended scallops were placed in 36 floating units (2 m below the surface), each consisting of three bags of 50, 100 and 200 scallops/bag. Scallops were deployed in March/April of 2005 and 2006. From May–September/October scallop abundances were monitored monthly and growth was quantified biweekly. Monthly survival was estimated by counting live free-planted scallops in 12–16 quadrats (1 m^2)/plot and counting live scallops/bag. Growth (shell height increase) was assessed for 20 scallops/methods/sampling date. In August 2005 and 2006, biofouling organisms growing on 156–159 scallop shells were scrapped and weighed.

A replicated, controlled study in 2009 in one area of seabed of Vancouver Island, North Pacific Ocean, Canada (4) found that transplanting hatchery-reared northern abalone *Haliotis kamtschatkana* into the wild reduced survivorship after seven days. Survivorship was lower in transplanted abalones (average survivorship: 0.58) compared to non-transplanted hatchery-reared abalone kept in tanks (average survivorship: 0.97–0.99). In 2009 a total of 1,680 hatchery-raised abalone (4.2–6.5 cm shell length) were used in a project assessing the survivorship of transplanted abalone. Seven groups of 20 tagged abalone were transplanted onto the seabed at 10 m intervals (9 m water depth). Seven days after transplanting, surviving abalone were searched for and counted during circular surveys (5 m radius around each of the transplant locations). Seven groups of 140 abalone were kept in hatchery tanks (not transplanted) for comparison. After seven days, the number of surviving abalone in tanks was determined.

A replicated, before-and-after study in 2005–2010 of 23 sites across five areas of in Peconic Bays, North Atlantic Ocean, New York, USA (5 - same expeimental set-up as 6) found that over four years after initiating transplantation of hatchery-reared bay scallop *Argopecten irradians irradians*, larval recruitment increased across all areas. Larval recruitment across all five areas was higher after restoration (2010: 29–118 spat/collector/day), compared to before (2005: 2–10 spat/collector/day), including two areas where no scallops had been transplanted, suggesting larval transport from restored sites to unrestored sites. A restoration programme aimed to increase scallop reproductive success was initiated in 2006 by transplanting several millions of hatchery-reared bay scallops in nets or directly on the seabed (100–200 scallops/m^2; see paper for details). Larval recruitment was monitored at 23 sites across five embayments (three with transplanted scallops, two nearby without to assess larval transport) for 6 years: 2005–2006 (before intensive restoration) and 2007–2010 (after commencement of intensive restoration). Spat collectors were deployed (3/site) at 1–6 m average depth before 1st June to sample bay scallop larvae. A second set of collectors was deployed three weeks later. Every three weeks thereafter, a new set of collectors replaced those that had been in the water for six weeks. After retrieval, all scallops in the spat collectors were counted and shell heights were measured.

A replicated, before-and-after study in 2005–2012 in seven areas of seabed in Peconic Bays, North Atlantic Ocean, New York, USA (6 - same expeimental set-up as 5) found that over five to six years after initiating transplantation of hatchery-reared bay scallop *Argopecten irradians irradians*, abundance of juvenile bay scallops typically increased, but not that of adult bay scallops. In five of seven areas (including one area where no scallops had been transplanted, suggesting larval transport from restored sites to unrestored sites), juvenile (<1-year-old) scallop abundance was higher after restoration (2011–2012: 0.07–2.8 scallops/m^2), compared to before (2005: 0.002–0.08 scallops/m^2). adult (>1-year-old) scallop abundance was statistically similar before (2005: 0.01–0.06 scallops/m^2) and after transplantation (2011–2012: 0.004–0.2 scallops/m^2 scallops/m^2). A restoration programme aimed to increase scallop reproductive success was initiated in 2006 by transplanting several millions of hatchery-reared bay scallops in nets or directly on the seabed (100–200 scallops/m^2; see paper for details). Juvenile and adult scallops were monitored annually in autumn at 23 sites across seven embayments (five with transplanted scallops, two nearby without to assess larval transport) for 8 years: 2005–2006 (before intensive restoration) and 2007–2012 (after commencement of intensive restoration). Divers counted all scallops within 2–4 transects (50 m^2)/site.

A replicated, randomized, controlled study in 2012 in one oyster reef area in Chesapeake Bay, North Atlantic Ocean, USA (7a) found that restoring oyster reefs by releasing hatchery-reared larvae of Eastern oyster *Crassostrea virginica* using a direct setting technique resulted in higher average initial spat (young oyster) settlement (2.4–8.4 spat/shell) compared to using a traditional remote technique (0.6–4.6). In addition, using the direct technique 61% of the settled spat survived the winter, resulting in higher spat abundance at the restored site (189/m^2) compared to an adjacent non-restored site (6/m^2). No comparison of survival was made with spat released using the traditional remote technique. Larvae were released in summer 2012. Direct setting consisted of placing twelve trays (32 x 24 x 15 cm) filled with 30 oyster shells in one area of oyster reef at 2–3 m depth, and releasing approximately 2 x 10^6 hatchery-reared Eastern oyster larvae directly over it. Remote setting consisted of adding approximately 10^4 larvae to two tanks, each with six spat-collector bags (55 x 20 x 1.5 cm) containing 20 shells each. Three days after larval release, five shells/tray or /bag were retrieved, and the number of spat/shell counted. After winter 2012/2013, spat were counted on 20 shells/tray for the direct technique. Spat on nearby non-restored reef were counted in six 24 x 36 cm quadrats.

A replicated, randomized, controlled study in 2012 and 2013 of twelve plots (trays) in one oyster reef area in Chesapeake Bay, North Atlantic Ocean, USA (7b) found that after release, hatchery-reared larvae of Eastern oyster *Crassostrea virginica* settled and survived equally on cleaned and natural oyster shells for a month. In 2012 and 2013, three days after release, initial spat settlement was similar on cleaned (2012: 8.4 spat/shell; 2013: 3.1) and natural shells (2012: 2.4; 2013: 4.9). After a month, the number of surviving spat was similar on cleaned (2012: 1.3; 2013: 2.9) and natural shells (2012: 1.0; 2013: 1.4). Twelve trays (32 x 24 x 15 cm) filled with 30 oyster shells were placed in one area of oyster reef at 2–3 m depth. Six contained cleaned shells, six contained natural shells. In summer 2012, 2 x 10^6 hatchery-reared Eastern oyster larvae were released using a direct setting technique. Shells were retrieved after 3 days (5/tray) and one month (20/tray), and the number of spat/shell counted. Shells were replaced afterwards. This was repeated in 2013.

(1) Wilbur A.E., Seyoum S., Bert T.M. & Arnold W.S. (2005) A genetic assessment of bay scallop (*Argopecten irradians*) restoration efforts in Florida's Gulf of Mexico coastal waters (USA). *Conservation Genetics*, 6, 111–122.
(2) Guest J.R., Todd P.A., Goh E., Sivalonganathan B.S. & Reddy K.P. (2008) Can giant clam (*Tridacna squamosa*) populations be restored on Singapore's heavily impacted coral reefs? *Aquatic Conservation: Marine and Freshwater Ecosyst*ems, 18, 570–579.
(3) Tettelbach S.T., Barnes D., Aldred J., Rivara G., Bonal D., Weinstock A., Fitzsimons-Diaz C., Thiel J., Cammarota M.C., Stark A. & Wejnert K. (2011) Utility of high-density plantings in bay scallop, *Argopecten irradians irradians,* restoration. *Aquaculture International*, 19, 715–739.
(4) Hansen S.C. & Gosselin L.A. (2013) Do predators, handling stress or field acclimation periods influence the survivorship of hatchery-reared abalone *Haliotis kamtschatkana* outplanted into natural habitats? *Aquatic Conservation: Marine and Freshwater Ecosystems,* 23, 246–253.
(5) Tettelbach S.T., Peterson B.J., Carroll J.M., Hughes S.W.T., Bonal D.M., Weinstock A.J., Europe J.R., Furman B.T. & Smith C.F. (2013) Priming the larval pump: Resurgence of bay scallop recruitment following initiation of intensive restoration efforts. *Marine Ecology Progress Series*, 478, 153–172.
(6) Tettelbach S.T., Peterson B.J., Carroll J.M., Furman B.T., Hughes S.W.T., Havelin J., Europe J.R., Bonal D.M., Weinstock A.J. & Smith C.F. (2015) Aspiring to an altered stable state: Rebuilding of bay scallop populations and fisheries following intensive restoration. *Marine Ecology Progress Series*, 529, 121–136.
(7a,b) Steppe C.N., Fredriksson D.W., Wallendorf L., Nikolov M. & Mayer R. (2016) Direct setting of *Crassostrea virginica* larvae in a tidal tributary: applications for shellfish restoration and aquaculture. *Marine Ecology Progress Series*, 546, 97–112.

13.2. Transplant/release captive-bred or hatchery-reared species in predator exclusion cages

- **One study** examined the effects of transplanting or releasing hatchery-reared species in predator exclusion cages on their wild populations. The study was in the North Pacific Ocean[1] (Canada).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Survival (1 study):** One replicated, controlled study the North Pacific Ocean[1] found that hatchery-reared abalone transplanted in predator exclusion cages had similar survivorship following release compared to those transplanted directly onto the seabed.

Background

Many populations of marine subtidal benthic invertebrate species have declined or been depleted due to the multiple threats they are under, such as habitat loss and overharvest (Airoldi *et al*. 2008; Hobday *et al*. 2000). To counteract this phenomenon, captive-bred or hatchery-reared marine subtidal benthic invertebrates can be transplanted or released at a site, either to introduce a species to a new site (where they did not historically occur), to reintroduce a species to a site (where they used to occur), or to enhance the population at a site where the species is already present by increasing its abundance (Hansen & Gosselin 2013). Following transplantation or release, initial mortality can be high, for instance due to stress and predation. To potentially increase survival, animals can be transplanted/released in exclusion cages to reduce the initial predation while animals are acclimating to their new environment (Hansen & Gosselin 2013).

When species transplantation/release was undertaken without predator exclusion cage, evidence has been summarised under "Species management – Transplant/release captive-bred or hatchery-reared species". Evidence for other related intervention is summarised under "Species management – Translocate species" and "Habitat restoration and creation – Translocate biogenic or habitat-forming (biogenic) species".

Airoldi L., Balata D. & Beck M.W. (2008) The gray zone: relationships between habitat loss and marine diversity and their applications in conservation. *Journal of Experimental Marine Biology and Ecology*, 366, 8–15.

Hansen S.C. & Gosselin L.A. (2013) Do predators, handling stress or field acclimation periods influence the survivorship of hatchery-reared abalone *Haliotis kamtschatkana* outplanted into natural habitats? *Aquatic Conservation: Marine and Freshwater Ecosystems,* 23, 246–253.

Hobday A.J., Tegner M.J. & Haaker P.L. (2000) Over-exploitation of a broadcast spawning marine invertebrate: decline of the white abalone. *Reviews in Fish Biology and Fisheries*, 10, 493–514.

A replicated, controlled study in 2009 in one area of seabed off Vancouver Island, North Pacific Ocean, Canada (1) found that hatchery-reared northern abalone *Haliotis kamtschatkana* transplanted into the wild in predator exclusion cages did not have higher survivorship following release compared to those transplanted directly onto the seabed. For the first seven days after transplantation, abalone in predator cages (not yet released) had higher survivorship (96%) than those not transplanted in cages (57%). However, seven days after being released from their cages, survivorship of abalone had decreased (42%) and was similar to those directly transplanted onto the seabed (34%). In addition, transplanting abalone in cages 1 m above the seabed or in cages onto the seabed led to similar survivorship, both before release (after 7 days; 96% vs 96%) and after release (after 14 days; 38% vs 46%). In 2009 a total of 1,680 hatchery-raised abalone (4.2–6.5 cm shell length) were used in a project assessing the survivorship of transplanted abalone. Three groups of 20 tagged abalone were transplanted at each of seven locations 10 m apart (9 m water depth). Each group corresponded to one of three treatments: 1) abalone placed in predator exclusion cages suspended 1 m above the seabed, 2) abalone placed in predator exclusion cages onto the seabed, 3) abalone transplanted directly onto the seabed (no cage). Seven days after transplanting, abalone in predator exclusion cages were released and allowed to disperse. On day 7 and 14 following transplanting, surviving abalone were searched for and counted inside all cages and during circular surveys (5 m radius around each of the transplantation locations).

(1) Hansen S.C. & Gosselin L.A. (2013) Do predators, handling stress or field acclimation periods influence the survivorship of hatchery-reared abalone *Haliotis kamtschatkana* outplanted into natural habitats? *Aquatic Conservation: Marine and Freshwater Ecosystems,* 23, 246–253.

13.3. Translocate species

Background

Many populations of marine subtidal benthic invertebrate species have declined or been depleted due to the multiple threats they are under, such as habitat loss and overharvest (Airoldi *et al*. 2008; Hobday *et al*. 2000). To counteract this phenomenon, marine subtidal benthic invertebrates can be translocated from a site with a healthy population, either to introduce a species to a new site (where they did not historically occur), to reintroduce a species to a site (where they used to occur), or to enhance the population at a site where the species is already present by increasing its abundance (Hughes *et al.* 2008; Swan *et al*. 2016). As the outcomes of translocating species can vary largely with the type of species, studies have been grouped by broader taxonomic group (e.g: crustaceans such as lobsters or prawns; molluscs such as abalone, scallops, or mussels; worms).

When translocation is undertaken for a habitat-forming (biogenic) species, effects on the invertebrates associated with the habitat are reported in "Habitat restoration and creation – Translocate habitat-forming (biogenic) species". Evidence from transplantation studies of hatchery-reared species is summarised under "Species management – Transplant/release captive-bred or hatchery-reared species" and "Habitat restoration and creation – Transplant/release captive-bred or hatchery-reared habitat-forming (biogenic) species".

Airoldi L., Balata D. & Beck M.W. (2008) The gray zone: relationships between habitat loss and marine diversity and their applications in conservation. *Journal of Experimental Marine Biology and Ecology*, 366, 8–15.

Hobday A.J., Tegner M.J. & Haaker P.L. (2000) Over-exploitation of a broadcast spawning marine invertebrate: decline of the white abalone. *Reviews in Fish Biology and Fisheries*, 10, 493–514.

Hughes D.J., Poloczanska E.S. & Dodd J. (2008) Survivorship and tube growth of reef-building *Serpula vermicularis* (Polychaeta: Serpulidae) in two Scottish sea lochs. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 18, 117–129.

Swan K.D., McPherson J.M., Seddon P.J. & Moehrenschlager A. (2016) Managing marine biodiversity: the rising diversity and prevalence of marine conservation translocations. *Conservation Letters*, 9, 239–251.

13.3.1. Translocate crustaceans

- **One study** examined the effects of translocating crustacean species on their wild populations. The study took place in the Tasman Sea[1] (Australia).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Crustacean survival (1 study):** One study in the Tasman Sea[1] found that following translocation survival of southern rock lobsters was similar to that of resident lobsters.

A study in 2005–2007 in one area of rocky reef off the coast of southeastern Tasmania, Tasman Sea, Australia (1) found that two years after southern rock lobsters

Jasus edwardsii were translocated, their survival was similar to that of resident lobsters. Survival of translocated lobsters was 96–98% after two years, similar to resident lobsters (98%). In 2005, lobsters were translocated from a site where lobsters grew slowly to a site inside a marine reserve where resident lobsters grew faster. Survival was monitored for two years. Lobsters (n=1,998) were caught in the slow-growth site using baited pots, tagged, and kept in flow-through tanks with ambient seawater until release into the new site 2–3 days later. At the surface, batches of 50 lobsters were released into a net connected to a cage on the seabed. After 24h, all lobsters were released. Lobsters residing in the fast-growth site (2,668 in total) were tagged and monitored for comparison. Translocated and resident lobsters were resampled nine times using 20–60 baited pots. A mark-recapture model based on the number of recaptured tagged lobsters (457 translocated and 797 resident lobsters in total) was used to estimate percentage survival.

(1) Green B.S. & Gardner C. (2009) Surviving a sea-change: survival of southern rock lobster (*Jasus edwardsii*) translocated to a site of fast growth. *ICES Journal of Marine Science,* 66, 656–664.

13.3.2. Translocate molluscs

- **Nine studies** examined the effects of translocating mollusc species on their wild populations. Three examined scallops in the North Atlantic Ocean[1a,b] (USA) and the Tasman Sea and South Pacific Ocean[2] (New Zealand). One study examined conch in the Florida Keys[3] (USA). One examined clams in the North Atlantic Ocean[4] (Portugal). One examined abalone in the North Pacific Ocean[5] (USA). One examined mussels in Strangford Lough[6] (UK). Two examined mussels in the Gulf of Corinth[7a,b] (Greece).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (8 STUDIES)

- **Mollusc abundance (3 studies):** One replicated, controlled, before-and-after study in the North Atlantic Ocean[1b] found that translocating bay scallops increased larval recruitment into the adult population compared to before translocation. One before-and-after study in the North Pacific Ocean[5] found that following translocation of adult pink abalone to existing patchy populations, total abalone abundance (translocated and resident) decreased to similar levels as before translocation. One replicated, site comparison study in Strangford Lough[6] found that after translocating horse mussels, the abundance of young mussels was higher in site with translocated mussels compared to both sites without translocated mussels and natural mussel reefs.
- **Mollusc reproductive success (1 study):** One replicated, controlled, before-and-after study in the North Atlantic Ocean[1b] found that translocating bay scallops did not increase larval production compared to before translocation.
- **Mollusc survival (5 studies):** Three replicated studies (one before-and-after and two site comparisons) in the North Atlantic Ocean[1a,4] and in the Tasman Sea and South Pacific Ocean[2], found that following translocation, scallops and clams survived. Survival of translocated New Zealand scallops[2] was higher in areas closed to commercial fishing compared to fished areas. Two studies in the Gulf of Corinth[7a,b] found that Mediterranean fan mussels survived when translocated to a deep site, and had similar survival compared to naturally-occurring mussels[7b], but did not survive when translocated to a shallow site[7a].
- **Mollusc condition (2 studies):** One replicated, site comparison study in the North Atlantic Ocean[4] found that following translocation, clams had similar condition indices to clams in the source site. One study in the Gulf of Corinth[7b] found that translocated Mediterranean fan mussels had similar size-specific growth-rates compared to naturally-occurring mussels.

BEHAVIOUR (1 STUDY)

- **Mollusc behaviour (1 study):** One replicated study in the Florida Keys[3] found that translocating non-reproductive adult queen conch to aggregations of reproductive conch did not have adverse effects on the movement patterns of non-translocated resident conch, and all conch displayed similar total distance travelled, movement rates, migration patterns, home-range sizes, and sociability.

A replicated, before-and-after study in 1992 of four sites of seagrass bed in Bogue Sound estuary, Northern Carolina, North Atlantic Ocean, USA (1a) found that up to six months after translocation, bay scallops *Argopecten irradians concentricus* survived at all sites. Following translocation, average scallop abundance (representative of survival) at the transplant sites did not significantly change (directly after: 8.0–10.3; 6 months after: 5.9–11.6/0.5 m^2) and remained higher than before translocation (0.3–3.6/0.5 m^2). In July 1992, adult bay scallops (135,000 in total) were translocated in coolers without water to sites with low scallop densities (0.7/m^2). Scallops were deposited in one 30 x 40 m marked area at each site. Bay scallops were counted in 2 m^2 quadrats (n=16–24) inside the marked area two weeks before and on five occasions after translocation ending in December.

A replicated, before-and-after, site comparison study in 1988–1994 of three to four seagrass bed sites in one to three estuarine locations in Northern Carolina, North Atlantic Ocean, USA (1b) found that translocating bay scallops *Argopecten irradians concentricus* did not increase larval production but increased recruitment into the adult population compared to before translocation. Larval production was similar before (6–195/sample) and after (5–15) translocation, and remained lower than at sites in other estuaries (55–335). Larval recruitment (as abundance of settled spat) increased by on average 568% at translocation sites, while recruitment at sites without translocation increased (non-significantly) by 34%. In 1992–1994, adult bay scallops (100,000–150,000/year) were translocated in coolers without water to sites in Bogue Sound with low scallop abundance within seagrass beds. Larvae collectors (8–20) were deployed in 1988, 1989, 1992 and 1993 at one translocation site and at two sites without translocation in nearby estuaries (Core Sound; Back Sound). Settled scallop larvae were counted for each collector after two months. In 1988, 1989, 1992, 1993, and 1994 at two translocation sites and the same two sites without translocation, 0.5 m^2 plots were dredged (35–61 plots/site) and scallops under 1-year old counted.

A replicated, site comparison study in 2001 of 12 sites across four areas of soft seabed in the Tasman Sea and South Pacific Ocean, New Zealand (2) found that, a week after translocation, >40% of translocated New Zealand scallop *Pecten novaezelandiae* had survived. In addition, mortality was lower in areas closed to commercial fishing compared to fished areas. Mortality in the two closed areas (15% and 24%) were lower compared to the two fished areas (39% and 59%) in three of four comparisons (24% not statistically different to 39%). In addition, across the two fished areas, mortality was significantly higher in the area also seeded with juvenile scallops (59%), than the area not seeded (39%). In 2001, scallops were translocated from nearby areas to two areas closed to commercial fishing (in February and April) and two fished areas (in April) (three sites/area). One of the fished areas had also been seeded with approximately 1.2 million juvenile scallops in February. At each site, a 4-m long chain was deployed, with 12 scallops attached at fixed intervals. The status (dead or alive) of all scallops was checked daily for three days and after a week, and mortality assessed.

A replicated study in 2001 of two sites of seagrass, coral rubbles, and sandy seabed in the Florida Keys, between the North Atlantic Ocean and the Gulf of Mexico, USA (3) found that non-reproductive adult queen conch *Strombus gigas* translocated to aggregations of reproductive conch typically displayed similar behaviour to non-translocated resident conch, but effects varied with sites. At Looe Key, there were no differences between translocated and resident conch in total distances travelled (translocated: 203 vs resident: 270 m), movement rates (1.2 vs 1.1 m/day), migration patterns (reported as an index), home-range sizes (13,900 vs 13,200 m^2), and conch-conch interactions (reported as a sociability coefficient). At Easter Sambo, there were no differences between translocated and resident conch in total distances travelled (186 vs 144 m), movement rates (1.2 vs 0.8 m/day), and migration patterns, but translocated conch had larger home-range sizes (30,300 m^2) than resident conch (3,700 m^2) and interacted more with other translocated conch than with resident conch. Authors suggested that differences in conch behaviour were associated with differences in habitats between sites. In 2001, non-reproductive adult queen conch were translocated from a near-shore site to two offshore sites in an enforced protected area with aggregations of reproductive adult queen conch (Eastern Sambo: 132 conch; Looe Key: 255 conch). Conch were tagged with acoustic transmitters and their movements followed bimonthly for 10 months (Eastern Sambo: six translocated, six resident; Looe Key: five translocated, five resident).

A replicated, site comparison study in 2003–2004 of two sites off the coast of Algarve, North Atlantic Ocean, southern Portugal (4) found that after translocation, clams *Spisuloa solida* were in similar condition to clams in the source site, and that despite increased mortality over time 45% survived up to a year. Before translocation, the condition of all clams was 6.3 (ratio without units). After three months, conditions were similar for translocated (6.4–6.5) and source site clams (7.4–7.8). Survival of translocated clams was 65-85% after two weeks, 52-60% after three months, and 45% after a year. Size of translocated clams did not affect survival or condition. In 2003, a total of 4,000 clams were translocated from a source site to two 50 m^2 plots in a depleted site (1 clam/m^2) inside an area closed to fishing. Each plot was sub-divided into fifty 1 m^2 subplots. Clams, divided into sublegal (<25 mm shell length) and legal (>25 mm) size groups, were equally distributed to each subplot. After two weeks and three months, all clams/subplot were counted, and the condition index of 10 clams/subplot assessed. Clam condition was also assessed for the source site. After a year, all clams in the translocation site were counted.

A before-and-after study in 2009–2010 in one area of kelp forest, in the North Pacific Ocean, off the coast of San Diego, California, USA (5) found that 18 months after translocation of adult pink abalone *Haliotis corrugata* to existing patchy populations, total abalone (translocated and resident) abundance had decreased to similar levels as before translocation. Results were not statistically tested. Eighteen months after translocation, 35% of abalone at the site had been lost (due to mortality and emigration). Abalone abundance after 18 months was 0.11 abalone/m^2, lower than immediately after translocation (0.18 abalone/m^2) and more similar to before translocation (0.09 abalone/m^2). Following translocation, translocated abalone displayed similar average home range (163 m^2) and linear distance travelled (7 m) compared to resident abalone (home range: 145 m^2; distance travelled: 8.6 m). The study site (254 m^2) had 23 resident adult abalone. In September 2009, all were tagged with acoustic transmitters and returned to their original position. An additional 23 abalone from a nearby area (approximately 2.2 km north) were tagged and translocated to the study site in groups of 2–6. Divers monitored abalone for 18 months by counting dead tagged abalone and live untagged abalone. Home range and linear distance travelled by tagged abalone were assessed from their behaviour and movement patterns.

A replicated, site comparison study in 2010–2011 of 10 sites in Strangford Lough, Northern Ireland, UK (6) found that over a year after translocating habitat-forming horse mussel *Modiolus modiolus*, the abundance of mussel spat (young mussels) was higher in site with translocated mussels compared to both sites without translocated mussels and natural mussel reefs. Sites with translocated mussels had more spat ($164/m^2$) compared to sites without ($0/m^2$), and compared to natural reefs ($6/m^2$). In 2010 divers translocated live adult horse mussels from nearby natural mussel patches within the Lough to four sites (1,000 mussels/sites). After 12 months, two quadrats (0.25 x 0.25 m) were deployed at each site with translocated mussels and at four adjacent natural sites without translocated mussels. Spat were counted from sediment and shell samples for each quadrat. Natural horse mussel communities from two nearby horse mussel reefs within the Lough were sampled in December 2010 using the same sampling methodology.

A study in 2006 of two sites of unspecified seabed in Lake Vouliagmeni, Gulf of Corinth, Greece (7a) found that up to a year after translocation, most Mediterranean fan mussels *Pinna nobilis* survived in a deep site, but none in a shallow site. In the shallow site, all mussels were dead after 72 days, mostly due to poaching (90%). In the deep site 80% of mussels survived, 100% of mussel death was natural, and 75% of dead mussels were small (<6 cm). No statistical tests were performed. During a pilot study in July 2006, forty mussels were manually uprooted from a shallow area of the lake (4 m depth), their shell width measured, and translocated equally back to that same area or a deeper area (12 m). Translocated mussels in both areas were 1 m apart. Mussel survival was monitored by divers and mortality classed as "poaching" or "natural", after 12 days, 72 days, and one year.

A study in 2007–2012 in one area of unspecified seabed in Lake Vouliagmeni, Gulf of Corinth, Greece (7b) found that translocated Mediterranean fan mussels *Pinna nobilis* had similar survival and growth rate compared to naturally-occurring mussels. After five years, the survival of translocated mussels (96%) was similar to that of naturally-occurring mussels (95%). Size-specific growth was similar in translocated (smallest: 39%; largest: 1.5%) and naturally-occurring mussels (smallest: 47%; largest: 0%). Data for other size-classes were not provided. In 2007, forty-five mussels were manually uprooted from a shallow area of the lake (4 m depth) and translocated to a deeper area (12 m depth) in five groups (20 m apart) of 9 mussels (0.5 m apart). Yearly for five years, translocated mussels' survival was monitored by divers and mortality classed as "poaching" or "natural". Their shell width was measured, and mussels categorised in one of six size-classes. Twenty naturally-occurring mussels occurring at 12 m depth were also monitored.

(1a,b) Peterson C.H., Summerson H.C.,& Luettich Jr.R.A. (1996) Response of bay scallops to spawner transplants: a test of recruitment limitation. *Marine Ecology Progress Series*, 132, 93–107.

(2) Talman S.G., Norkko A., Thrush S.F. & Hewitt J.E. (2004) Habitat structure and the survival of juvenile scallops *Pecten novaezelandiae*: Comparing predation in habitats with varying complexity. *Marine Ecology Progress Series*, 269, 197–207.
(3) Delgado G.A. & Glazer R.A. (2007) Interactions between translocated and native queen conch *Strombus gigas*: evaluating a restoration strategy. *Endangered Species Research*, 3, 259–266.
(4) Joaquim S., Gaspar M.B., Matias D., Ben-Hamadou R. & Arnold W.S. (2008) Rebuilding viable spawner patches of the overfished *Spisula solida* (Mollusca: Bivalvia): a preliminary contribution to fishery sustainability. *ICES Journal of Marine Science*, 65, 60–64.
(5) Coates J.H., Hovel K.A., Butler J.L., Peter Klimley A. & Morgan S.G. (2013) Movement and home range of pink abalone *Haliotis corrugata*: Implications for restoration and population recovery. *Marine Ecology Progress Series*, 486, 189–201.
(6) Fariñas-Franco J.M., Allcock L., Smyth D. & Roberts D. (2013) Community convergence and recruitment of keystone species as performance indicators of artificial reefs. *Journal of Sea Research,* 78, 59–74.
(7a,b) Katsanevakis S. (2016) Transplantation as a conservation action to protect the Mediterranean fan mussel *Pinna nobilis*. *Marine Ecology Progress Series*, 546, 113–122.

13.3.3. Translocate worms

- **One study** examined the effects of translocating worm species on their wild populations. The study was in Scottish Lochs[1] (UK).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Worm survival (1 study):** One replicated, controlled study in Scottish Lochs[1] found that no reef-forming red tube worm survived when translocated to a new Loch, but survival was high when worms were translocated back to its source Loch.
- **Worm condition (1 study):** One replicated, controlled study in Scottish Lochs[1] found that no reef-forming red tube worm survived and so no growth was recorded when translocated to a new loch, worms translocated back to its source Loch grew.

A replicated, controlled study in 2004–2005 of four soft seabed sites in two sea Lochs in west Scotland, UK (1) found that a year after translocation survivorship and growth of the reef-forming red tube worm *Serpula vermicularis* were different when translocated to a new Loch or back to the source Loch. In Loch Sween (new Loch), translocated tubes gradually disappeared and only remnants remained after one year. No mortality and growth data were recorded for Loch Sween. In Loch Creran (the source Loch) 76% of tubes were recovered after one year. Mortality averaged 5.3% and tube growth averaged 32–33 mm/year at Loch Creran. In June and July 2004, clusters of tubes containing living individuals of the red tube worm were manually collected by divers from one site in Loch Creran. Seven to ten days later, 10 clusters (10 tubes with living worm/cluster) were translocated at 1 m intervals to each of four sites: two in Loch Sween where wild populations died out (3–4 m depth) and two back in Loch Creran (9–10 m depth). For a year, clusters were monitored at intervals, and the remaining clusters recovered in July 2005. For each tube, the presence of living worm was recorded, and its growth measured.

(1) Hughes D.J., Poloczanska E.S. & Dodd J. (2008) Survivorship and tube growth of reef-building *Serpula vermicularis* (Polychaeta: Serpulidae) in two Scottish sea lochs. *Aquatic Conservation: Marine and Freshwater Ecosystems,* 18, 117–129.

13.4. Provide artificial shelters following release

- We found no studies that evaluated the effects of providing artificial shelters following the release of species 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

Many populations of marine subtidal benthic invertebrate species have declined or been depleted due to the multiple threats they are under, such as habitat loss and overharvest (Airoldi *et al*. 2008; Hobday *et al*. 2000). To counteract this phenomenon, captive-bred (or hatchery-reared) marine subtidal benthic invertebrates can be transplanted or released at a site, either to introduce a species to a new site (where they did not historically occur), to reintroduce a species to a site (where they used to occur), or to enhance the population at a site where the species is already present by increasing its abundance (Hansen & Gosselin 2013). To potentially increase the animals' survival following transplant/release, some assistance can be provided, such as providing artificial shelters for the animals to avoid predation and settle into their new environment (Gutzler *et al*. 2015).

Related evidence is summarised under "Species management – Transplant/release captive-bred or hatchery-reared species", "Transplant/release captive-bred or hatchery-reared species in predator exclusion cages", "Translocate species", and "Provide artificial shelters".

Airoldi L., Balata D. & Beck M.W. (2008) The gray zone: relationships between habitat loss and marine diversity and their applications in conservation. *Journal of Experimental Marine Biology and Ecology*, 366, 8–15.

Gutzler B.C., Butler M.J. & Behringer D.C. (2015) Casitas: A location-dependent ecological trap for juvenile Caribbean spiny lobsters, *Panulirus argus*. *ICES Journal of Marine Science,* 72, 177–184.

Hansen S.C. & Gosselin L.A. (2013) Do predators, handling stress or field acclimation periods influence the survivorship of hatchery-reared abalone Haliotis kamtschatkana outplanted into natural habitats? *Aquatic Conservation: Marine and Freshwater Ecosystems,* 23, 246–253.

Hobday A.J., Tegner M.J. & Haaker P.L. (2000) Over-exploitation of a broadcast spawning marine invertebrate: decline of the white abalone. *Reviews in Fish Biology and Fisheries*, 10, 493–514.

13.5. Set recreational catch quotas

- We found no studies that evaluated the effects of setting recreational catch quotas on subtidal benthic invertebrate 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

Many populations of marine subtidal benthic invertebrate species have declined or been depleted due to the multiple threats they are under, including overharvest (Hobday *et al*. 2000). Populations of certain species have decline to such extent that they are now protected, and their fishing and harvest is controlled and/or illegal (Stierhoff *et al*. 2012). Recreational fishing and harvest quotas (such as Total Allowable Catch) are a means by which many governments and local regulatory bodies regulate biological resources (species stocks). Catch quotas are limits, expressed for instance in weight or numbers of animals, that one person or vessel is allowed to fish or harvest over a certain period. Setting catch quotas for specific species, particularly those declining or endangered, can potentially reduce the pressure from fishing and harvesting, and allow the species to recover over time (Lewis 2015).

Related evidence is summarised under "Threat: Biological resource use – Set commercial catch quotas".

Hobday A.J., Tegner M.J. & Haaker P.L. (2000) Over-exploitation of a broadcast spawning marine invertebrate: decline of the white abalone. *Reviews in Fish Biology and Fisheries*, 10, 493–514.

Lewis S.G. (2015) Bags and tags: randomized response technique indicates reductions in illegal recreational fishing of red abalone (*Haliotis rufescens*) in Northern California. *Biological Conservation,* 189, 72–77.

Stierhoff K.L., Neuman M. & Butler J.L. (2012) On the road to extinction? Population declines of the endangered white abalone, *Haliotis sorenseni*. *Biological Conservation*, 152, 46–52.

13.6. Establish size limitations for the capture of recreational species

- We found no studies that evaluated the effects of establishing size limitations for the capture of recreational species on subtidal benthic invertebrate 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

Many populations of marine subtidal benthic invertebrate species have declined or been depleted due to the multiple threats they are under, including overharvest (Hobday *et al*. 2000). Populations of certain species have decline to such extent that they are now protected (Stierhoff *et al*. 2012), and their fishing and harvest is controlled, for instance by setting maximum and minimum limits on the size (usually correlated with age) of animals allowed to be caught (Van Poorten *et al*. 2013). Setting minimum size limits can protect juveniles and animals that have not yet reached sexual maturity, potentially allowing them to reach adulthood and reproduce. Setting maximum size limits can protect older mature animals, which often contribute more strongly to reproduction and population renewal (for instance older females usually produce more or bigger eggs).

13.7. Tag species to prevent illegal fishing or harvesting

- **One study** examined the effects of tagging species to prevent illegal fishing or harvesting on subtidal benthic invertebrates. The study examined the effects on the Californian abalone fishery[1] (USA).

COMMUNITY RESPONSE (0 STUDIES)
POPULATION RESPONSE (0 STUDIES)
BEHAVIOURS (1 STUDY)

- **Behaviour-change (1 study):** One before-and-after study in California[1] found no significant reduction in non-compliance with daily quotas of abalones after introducing tagging regulations.

OTHER (1 STUDY)

- **Illegal catch (1 study):** One before-and-after study in California[1] found no significant reduction in illegal takes of abalones after introducing tagging regulations.

Background

Many populations of marine subtidal benthic invertebrate species have declined or been depleted due to the multiple threats they are under, including overharvest (Hobday *et al*. 2000). Populations of certain species have declined to such extent that they are now protected, and their fishing and harvesting is controlled and/or illegal (Stierhoff *et al*. 2012). For species for which fishing or harvesting is forbidden, animals can be tagged to deter illegal capture and potentially help reduce their population declines (Lewis 2015).

Evidence for related interventions is summarised under "Threat: Biological resource use – Set commercial catch quotas", and "Species management – Set recreational catch quotas" and "Etablish size limitations for the capture of recreational species".

Hobday A.J., Tegner M.J. & Haaker P.L. (2000) Over-exploitation of a broadcast spawning marine invertebrate: decline of the white abalone. *Reviews in Fish Biology and Fisheries*, 10, 493–514.

Lewis S.G. (2015) Bags and tags: randomized response technique indicates reductions in illegal recreational fishing of red abalone (*Haliotis rufescens*) in Northern California. *Biological Conservation,* 189, 72–77.

Stierhoff K.L., Neuman M. & Butler J.L. (2012) On the road to extinction? Population declines of the endangered white abalone, *Haliotis sorenseni*. *Biological Conservation*, 152, 46–52.

A before-and-after study in 2007 and 2011 of fishers surveyed across 11 sites in northern California, USA (1) found that introducing tagging regulation did not reduce overall illegal takes of red abalone *Haliotis rufescens*. Tagging led to a 4% reduction in illegal takes of abalone, but this was not statistically significant. Of the seven categories of illegal takes considered, only non-compliance with daily-take quotas significantly reduced (before tagging: 32%; after tagging: 19%), particularly amongst local fishers (before: 72%; after: 43%). The other six categories were not significantly reduced (see paper for details). Red abalone tagging regulation was introduced in California between 2007 and 2011 (date unspecified). Over five weeks in August–September 2007 and 2011, fishers at 11 sites where abalone harvest is restricted were asked to respond to a set questionnaire regarding their compliance to each of seven regulations. Proportional non-compliance across fishers was estimated for each regulation and overall.

(1) Lewis S.G. (2015) Bags and tags: randomized response technique indicates reductions in illegal recreational fishing of red abalone (*Haliotis rufescens*) in Northern California. *Biological Conservation*, 189, 72–77.

13.8. Cease or prohibit the harvest of scallops

- **Three studies** examined the effects of ceasing or prohibiting the harvest of scallops on their populations. One study was in the South Atlantic Ocean[1] (Argentina), one in the English Channel[2] (UK) and one in the Irish Sea[3] (UK).

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (3 STUDIES)

- **Scallop abundance (3 studies):** Two of three site comparison studies (one replicated, one before-and-after) in the South Atlantic Ocean[1], the English Channel[2], and the Irish Sea[3] found that in areas where scallop harvesting had stopped scallop abundance was similar[2,3] and scallop biomass[1] higher compared to harvested areas.

Background

Commercial harvest of scallops is usually undertaken using dredges, and as such can impact subtidal benthic invertebrates through removal of untargeted species and damage to the seabed. Recreational and artisanal scallop fishing may cause less impact due to the smaller scale of the operations and the harvesting methods used (for instance hand-harvest). Nevertheless, both harvest methods have negative consequences on scallop populations, particularly if harvesting levels are unsustainable. Bylaws, legislation, or voluntary agreements could be established which prohibit the harvest of scallops in an area, potentially allowing scallop populations to recover over time (Bull 1989; Schejter *et al.* 2008).

When this intervention occurs within a protected area, evidence has been summarised under "Habitat protection – Designate a Marine Protected Area and cease or prohibit harvest of scallops". Evidence for related interventions is summarised under "Threat: Biological resource use – Cease or prohibit scallops dredging" and "Habitat protection – Designate a Marine Protected Area and prohibit dredging".

Bull M.F. (1989) *The New Zealand scallop fishery: a brief review of the fishery and its management.* Pages 42. Ed. MLC Dredge, WF Zacharin & LM Joli.

Schejter L., Bremec C.S. & Hernández D. (2008) Comparison between disturbed and undisturbed areas of the Patagonian scallop (*Zygochlamys patagonica*) fishing ground "Reclutas" in the Argentine Sea. *Journal of Sea Research*, 60, 193–200.

A site comparison study in 1998–2002 in two soft seabed areas in the South Atlantic Ocean, Argentina (1) found that an area prohibiting the commercial dredging of Patagonian scallops *Zygochlamys patagonica* had a higher biomass of scallops compared to adjacent fished areas. Six years after closure, the biomass of scallops was higher in the closed area (4–12 kg/100 m^2), compared to the fished area (1–10 kg/100 m^2). The area was closed to commercial dredging of scallops in 1996. Samples were collected at 100 m depth once a year in 1998–2002 using a dredge (generalist dredge not specifically targeting scallops; 10 mm mesh) at 23 sites in the closed area and at 71 adjacent sites outside. Scallops were weighed and counted. Information was updated using an erratum (Schejter *et al.*, 2009).

Schejter L., Bremec C.S. & Hernández D. (2009) Erratum to "Comparison between disturbed and undisturbed areas of the Patagonian scallop (*Zygochlamys patagonica*) fishing ground "Reclutas" in the Argentine Sea" [J. Sea Research 60/3 (2008) 193]. *Journal of Sea Research* 61, 275.

A replicated, site comparison study in 2007 in six rocky seabed areas in Lyme Bay, English Channel, UK (2) found that areas closed to commercial scallop dredging did not have higher abundances of king scallop *Pecten maximus* or queen scallop *Aequipecten opercularis*, compared to areas which remained dredged. There was no significant difference in abundance between closed and dredged areas for king scallops (closed: 25–38; dredged: 27–28 individuals/100 m^2) or queen scallops (closed: 41–41; dredged: 80–97 individuals/100 m^2). In March and August 2007, six areas within the bay were sampled: three voluntarily closed to scallop dredging since September 2006 (but where static gear fisheries occurred) and three that remained open to scallop dredging. Samples were taken using a video camera (10 recordings/area) towed for approximately 10 minutes in a straight line. Abundances of each scallop species were recorded from the videos.

A before-and-after, site comparison study 2009–2011 in two areas of sandy, pebbly and gravelly seabed in Cardigan Bay, Irish Sea, Wales, UK (3) found that two years after prohibiting commercial scallop dredging year-round in an area, abundances of king scallop *Pecten maximus* and queen scallop *Aequipecten opercularis* had not increased and remained similar to that of an adjacent seasonally dredged area. Abundances of king and queen scallops were similar between closed and fished areas both before (king: closed 0.9 vs fished 0.8; queen: 0.2 vs 0.7 individuals/m^2) and two years after closure (king: 0.3 vs 0.3; queen: 0.2 vs 0.7 individuals/m^2). Two areas of Cardigan Bay were assessed: one permanently closed to scallop dredging in March 2010, and the other seasonally closed to scallop dredging (May to October). Surveys were conducted before closure (December 2009) and three times after (June 2010 to April 2011). During each survey, a camera was towed for 300 m at six sites/area (at 30 m depth). More than 40 images/camera tow (covering a 0.13 m^2 area of seabed) were analysed, and scallops were identified and counted.

(1) Schejter L., Bremec C.S. & Hernández D. (2008) Comparison between disturbed and undisturbed areas of the Patagonian scallop (*Zygochlamys patagonica*) fishing ground "Reclutas" in the Argentine Sea. *Journal of Sea Research*, 60, 193–200.
(2) Hinz H., Tarrant D., Ridgeway A., Kaiser M.J. & Hiddink J.G. (2011) Effects of scallop dredging on temperate reef fauna. *Marine Ecology Progress Series*, 432, 91–102.
(3) Sciberras M., Hinz H., Bennell J.D., Jenkins S.R., Hawkins S.J. & Kaiser M.J. (2013) Benthic community response to a scallop dredging closure within a dynamic seabed habitat. *Marine Ecology Progress Series,* 480, 83–98.

13.9. Cease or prohibit the harvest of conch

- We found no studies that evaluated the effects of ceasing or prohibiting the harvest of conch 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

Conch populations have significantly declined due to overharvest, for commercial and recreational purposes (Theile 2001). Bylaws, legislation, or voluntary agreements could be established which prohibit the harvest of conch in an area, in order to reduce the pressure on conch populations and in theory allow the population to recovery naturally.

When this intervention occurs in a marine protected area, the evidence is summarised under "Habitat protection – Designate a Marine Protected Area and prohibit harvest of conch".

Theile S. (2001) *Queen conch fisheries and their management in the Caribbean*. Brussels: TRAFFIC Europe.

13.10. Cease or prohibit the harvest of sea urchins

- We found no studies that evaluated the effects ceasing or prohibiting the harvest of sea urchins 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

Sea urchins can represent key species within a marine system, with other species crucially depending on their presence to thrive (Coyer *et al.* 1993). Commercial, but also recreational, harvest of edible sea urchins has led to significant changes, with for instance species of protected abalones suffering as a ripple effect (Rogers-Bennett & Pearse 2001). Bylaws, legislation, or voluntary agreements could be established which prohibit the harvest of sea urchins in an area, in theory allowing their population to recover, and as a consequence the wider subtidal benthic invertebrate communities which depend on them (Rogers-Bennett & Pearse 2001).

When this intervention occurs in a marine protected area, the evidence is summarised under "Habitat protection – Designate a Marine Protected Area and prohibit harvest of sea urchins".

Coyer J.A., Ambrose R.F., Engle J.M. & Carroll J.C. (1993) Interactions between corals and algae on a temperate zone rocky reef: mediation by sea urchins. *Journal of Experimental Marine Biology and Ecology,* 167, 21–37.

Rogers-Bennett L. & Pearse J.S. (2001) Indirect benefits of marine protected areas for juvenile abalone. *Conservation Biology*, 15, 642–647.

13.11. Remove and relocate invertebrate species before onset of impactful activities

- We found no studies that evaluated the effects of removing and relocating invertebrate species before onset of impactful activities 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

Many populations of marine subtidal benthic invertebrate species have declined or been depleted due to the multiple threats they are under, including habitat damage or loss and direct physical damages from anthropogenic activities (Airoldi *et al*. 2008; Hobday *et al*. 2000). As a pre-emptive conservation measure, species can be temporarily removed to allow for an impactful activity to occur, then relocated back into their original location, or at a different location. Such measure has been trialled to preserve maërl (rhodolith) species and the habitat it forms during the dredging of new shipping channels (Sheehan *et al*. 2015; see "Habitat restoration and creation – Remove and relocate habitat-forming (biogenic) invertebrate species before onset of impactful activities").

Airoldi L., Balata D. & Beck M.W. (2008) The gray zone: relationships between habitat loss and marine diversity and their applications in conservation. *Journal of Experimental Marine Biology and Ecology*, 366, 8–15.

Hobday A.J., Tegner M.J. & Haaker P.L. (2000) Over-exploitation of a broadcast spawning marine invertebrate: decline of the white abalone. *Reviews in Fish Biology and Fisheries*, 10, 493–514.

Sheehan E.V., Bridger D., Cousens S.L. & Attrill M.J. (2015) Testing the resilience of dead maerl infaunal assemblages to the experimental removal and re-lay of habitat. *Marine Ecology Progress Series*, 535, 117–128.

14. Education and awareness

Background

Compared to other groups of marine species, such as marine mammals, sharks, or corals, marine invertebrates (excl. corals) tend to be small and inconspicuous animals which attract less attention from the general public, despite some being of high economic and ecological values. This means that there are significant challenges in terms of gaining understanding or involvement from the public for marine invertebrate conservation.

Improving education and awareness of marine invertebrates, especially those under considerable pressures and threats, is crucial to achieve conservation objectives. This could be achieved through educational courses, workshops and outreach events, but also by integrating education and conservation within school curricula, businesses and recreational activities (for instance wildlife-spotting tours (Zeppel 2008). A number of conservation and citizen science programmes have attempted to raise awareness of marine conservation issues to the public, and such endeavours are increasing in popularity worldwide (Cigliano *et al.* 2015; Devictor *et al*. 2010). Ideally a quantitative change in behaviour, or effects on marine biodiversity would be measured within the project. However, such data are often not collected, particularly when it comes to marine subtidal benthic invertebrates. This synopsis therefore presents little evidence of the impacts of education and awareness on marine subtidal benthic invertebrates, but this does not mean such projects do not exist or are not beneficial. Furthermore, as Conservation Evidence does not systematically search journals and reports specialised in behavioural and social sciences, we are likely to have missed a considerable number of relevant studies in the field of education and awareness that are published in them.

Cigliano J.A., Meyer R., Ballard H.L., Freitag A., Phillips T.B. & Wasser A. (2015) Making marine and coastal citizen science matter. *Ocean & Coastal Management*, 115, 77–87.

Devictor V., Whittaker R.J. & Beltrame C. (2010) Beyond scarcity: citizen science programmes as useful tools for conservation biogeography. *Diversity and distributions*, 16, 354–362.

14.1. Provide educational or other training programmes about the marine environment to improve behaviours towards marine invertebrates

- **One study** examined the effects of providing educational or other training programmes about the marine environment on subtidal benthic invertebrate populations. The study took place in Hong Kong[1].

COMMUNITY RESPONSE (0 STUDIES)
POPULATION RESPONSE (0 STUDIES)
BEHAVIOUR (1 STUDY)

- **Behaviour change (1 study):** One replicated, before-and-after survey study in Hong Kong[1] found that a conservation education program on the Asian horseshoe crab in secondary schools significantly increased the students' behaviour towards Asian horseshoe crab conservation.

Background

Whether through school curricula, education programmes, training courses, workshops, or outreach events, providing education and training programmes can help raise awareness about marine conservation issues and potentially induce a behavioural change (Colleton *et al*. 2016; Kwan *et al.* 2017; Leisher *et al.* 2012). These programmes may be about the marine environment in general or about specific ecosystems, species, or groups of species, their biodiversity, conservation, and management.

Evidence for related interventioms is summarised under "Education and awareness – Organise educational marine wildlife tours to improve behaviours towards marine invertebrates".

Colleton N., Lakshman V., Flood K., Birnbaum M., Mcmillan K. & Lin A. (2016) Concepts and practice in the emerging use of games for marine education and conservation. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 26, 213–224.

Kwan B.K., Cheung J.H., Law A.C., Cheung S.G. & Shin P.K. (2017) Conservation education program for threatened Asian horseshoe crabs: a step towards reducing community apathy to environmental conservation. *Journal for Nature Conservation*, 35, 53–65.

Leisher C., Mangubhai S., Hess S., Widodo H., Soekirman T., Tjoe S., Wawiyai S., Larsen S.N., Rumetna L., Halim A. & Sanjayan M. (2012) Measuring the benefits and costs of community education and outreach in marine protected areas. *Marine Policy*, 36, 1005–1011.

A replicated, before-and-after study in 2009–2016 of 96 secondary schools in Hong Kong (1) found that a 14-month-long conservation education programme improved students' behaviour towards Asian horseshoe crab *Tachypleus tridentatus* conservation. The programme increased students' behaviour towards horseshoe crab conservation by 21%. This included a 43% increase in students promoting horseshoe crab conservation to relatives and friends, a 5% increase in students themselves promoting horseshoe crab conservation, and a 15% increase in their willingness-to-pay for conserving Asian horseshoe crabs. The programme also improved their general biology and ecology knowledge of Asian horseshoe crabs by 26% and their perception and awareness towards horseshoe crab conservation by 17%. Between 2009 and 2016, the "Juvenile Horseshoe Crab Rearing Program" took place in 96 schools. Teams of students reared juvenile crabs for 14 months before releasing them into nursery grounds. Before the start and after the end of each programme, students were asked to respond to a set questionnaire regarding their behaviour towards horseshoe crab conservation, knowledge of horseshoe crab biology/ecology and their perception and awareness of horseshoe crab conservation. A total of 1,391 students responded.

(1) Kwan B.K., Cheung J.H., Law A.C., Cheung S.G. & Shin P.K. (2017) Conservation education program for threatened Asian horseshoe crabs: a step towards reducing community apathy to environmental conservation. *Journal for Nature Conservation*, 35, 53–65.

14.2. Organise educational marine wildlife tours to improve behaviours towards marine invertebrates

- We found no studies that evaluated the effects of organising educational marine wildlife tours to induce behavioural changes and increase engagement in marine conservation on human behaviour and/or subtidal benthic invertebrate 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

Marine wildlife tours can be highly educational for visitors and provide conservation benefits (Zeppel & Muloin 2008). This has in part been linked to emotional and affective responses. Visitors can also learn about the biology, ecology, and behaviour of marine species; best practice guidelines; and human threats to marine life, during educational marine wildlife tours. Studies have shown that visitors who attended the tours wanted to be educated about marine issues (Lück 2003). Such tours have been linked to behavioural and intentional changes, increased empathy towards marine wildlife and increased engagement in marine conservation actions (Zeppel 2008).

Lück M. (2003) Education on marine mammal tours as agent for conservation—but do tourists want to be educated? *Ocean & Coastal Management*, 46, 943–956.

Zeppel H. & Muloin S. (2008) Conservation benefits of interpretation on marine wildlife tours. *Human Dimensions of Wildlife*, 13, 280–294.

Zeppel H. (2008) Education and conservation benefits of marine wildlife tours: Developing free-choice learning experiences. *The Journal of Environmental Education*, 39, 3–18.

References

Abate R.S. (2010) NEPA, national security, and ocean noise: the past, present, and future of regulating the impact of navy sonar on marine mammals. *Journal of International Wildlife Law & Policy*, 13, 326–356.

Abbott J.K. & Haynie A.C. (2012) What are we protecting? Fisher behavior and the unintended consequences of spatial closures as a fishery management tool. *Ecological Applications*, 22, 762–777.*

Acheson J. (1998) Lobster trap limits: A solution to a communal action problem. *Human Organization*, 57, 43–52.

Acosta C. & Robertson D. (2003) Comparative spatial ecology of fished spiny lobsters *Panulirus argus* and an unfished congener *P. guttatus* in an isolated marine reserve at Glover's Reef atoll, Belize. *Coral Reefs,* 22, 1–9.*

Acosta C.A. (2002) Spatially explicit dispersal dynamics and equilibrium population sizes in marine harvest refuges. *ICES Journal of Marine Science,* 59, 458–468. *

Adams T.P., Miller R.G., Aleynik D. & Burrows M.T. (2014) Offshore marine renewable energy devices as stepping stones across biogeographical boundaries. *Journal of Applied Ecology*, 51, 330–338.

Agnalt A.-L. (2008) Fecundity of the European lobster (*Homarus gammarus*) off southwestern Norway after stock enhancement: do cultured females produce as many eggs as wild females? *ICES Journal of Marine Science*, 65, 164–170.*

Agnalt A.-L. Kristiansen T.S. & Jørstad K.E. (2007) Growth, reproductive cycle, and movement of berried European lobsters (*Homarus gammarus*) in a local stock off southwestern Norway. *ICES Journal of Marine Science,* 64, 288–297.*

Aguado-Giménez F., Piedecausa M.A., Carrasco C., Gutiérrez J.M., Aliaga V. & García-García B. (2011) Do benthic biofilters contribute to sustainability and restoration of the benthic environment impacted by offshore cage finfish aquaculture? *Marine Pollution Bulletin*, 62, 1714–1724.*

Aguado-Giménez F., Piedecausa M.A., Gutiérrez J.M., García-Charton J.A., Belmonte A. & García-García B. (2012) Benthic recovery after fish farming cessation: a "beyond-BACI" approach. *Marine Pollution Bulletin*, 64, 729–738.*

Airoldi L., Balata D. & Beck M.W. (2008) The gray zone: relationships between habitat loss and marine diversity and their applications in conservation. *Journal of Experimental Marine Biology and Ecology*, 366, 8–15.

Airoldi L., Connell S.D. & Beck M.W. (2009) The loss of natural habitats and the addition of artificial substrata. Pages 269–280 in: Wahl M. (eds) *Marine Hard Bottom Communities*. Springer, Berlin, Heidelberg.

Alcock T.M. (1992) Ecology tankers and the Oil Pollution Act of 1990: A history of efforts to require double hulls on oil tankers. *Ecology Law Quaterly.* 19, 97.

Alexander T.J. (2013) Cryptic invertebrates on subtidal rocky reefs vary with microhabitat structure and protection from fishing. *Marine Ecology Progress Series*, 481, 93–104.*

Alexander T.J., Barrett N., Haddon M. & Edgar G. (2009) Relationships between mobile macroinvertebrates and reef structure in a temperate marine reserve. *Marine Ecology Progress Series,* 389, 31–44.*

Allendorf F.W. & Lundquist L.L. (2003) Introduction: population biology, evolution, and control of invasive species. *Conservation Biology*, 17, 24–30.

Al-Majed A.A., Adebayo A.R. & Hossain M.E. (2012) A sustainable approach to controlling oil spills. *Journal of Environmental Management*, 113, 213–227.

Althaus F., Williams A., Schlacher T.A., Kloser R.J., Green M.A., Barker B.A., Bax N.J., Brodie P. & Schlacher-Hoenlinger M.A. (2009) Impacts of bottom trawling on deep-coral ecosystems of seamounts are long-lasting. *Marine Ecology Progress Series,* 397, 279–294.*

Alzieu C. (2000) Impact of tributyltin on marine invertebrates. *Ecotoxicology*, 9, 71–76.

Ambrose R.F. & Anderson T.W. (1990). Influence of an artificial reef on the surrounding infaunal community. *Marine Biology,* 107, 41–52.

Amengual-Ramis J.F., Vazquez-Archdale M., Canovas-Perez C. & Morales-Nin B. (2016) The artisanal fishery of the spiny lobster *Palinurus elephas* in Cabrera National Park, Spain: comparative study on traditional and modern traps with trammel nets. *Fisheries Research*, 179, 23–32.*

Anderson R. (2005). *Environmental effects of marine finfish aquaculture (Vol. 5).* Springer Science & Business Media.

Andrady A.L. (2015) Persistence of plastic litter in the oceans. In *Marine anthropogenic litter* (57–72). Springer, Cham.

Andréfouët S., Van Wynsberge S., Gaertner-Mazouni N., Menkes C., Gilbert A. & Remoissenet G. (2013) Climate variability and massive mortalities challenge giant clam conservation and management efforts in French Polynesia atolls. *Biological Conservation*, 160, 190–199.*

Andriahajaina & Hockley (2007) The potential of native species aquaculture to achieve conservation objectives: freshwater crayfish in Madagascar. *The International Journal of Biodiversity Science and Management,* 3, 217–222.

Angel D.L., Eden N., Breitstein S., Yurman A., Katz T. & Spanier E. (2002) In situ biofiltration: a means to limit the dispersal of effluents from marine finfish cage aquaculture. *Hydrobiologia*, 469, 1–10.*

Arce A.M., Aguilar-Dávila W., Sosa-Cordero E. & Caddy J.F. (1997) Artificial shelters (casitas) as habitats for juvenile spiny lobsters *Panulirus argus* in the Mexican Caribbean. *Marine Ecology Progress Series*, 158, 217–224.

Arechavaia-Lopez P., Sanchez-Jerez P., Bayle-Sempere J.T., Uglem I. & Mladineo I. (2013) Reared fish. Farmed escapees and wild fish stockes – a triangle of pathogen transmission of concern to Mediterranean aquaculture management. *Aquaculture Environment Interactions*, 3, 153–161.

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.

Arrasate-López M., Tuset V.M., Santana J.I., García-Mederos A., Ayza O. & González J.A. (2012) Fishing methods for sustainable shrimp fisheries in the Canary Islands (North-West Africa). *African Journal of Marine Science*, 34, 331–339.*

Arthur J.R., Bondad-Reantaso M.G., Campbell M.L., Hewitt C.L., Phillips M.J. & Subasinghe R.P. (2009). Understanding and applying risk analysis in aquaculture: a manual for decision-makers. FAO Fisheries and Aquaculture Technical Paper No. 519/1. FAO; Rome. 113pp. doi: 10.13140/RG.2.1.2045.7207

Ashley M.C., Mangi S.C. & Rodwell L.D. (2014) The potential of offshore windfarms to act as marine protected areas–a systematic review of current evidence. *Marine Policy*, 45, 301–309.

Babcock R.C., Kelly S., Shears N.T., Walker J.W. & Willis T.J. (1999) Changes in community structure in temperate marine reserves. *Marine Ecology Progress Series,* 189, 125–134.*

Badalamenti F., Chemello R., D'anna G., Ramos P. H. & Riggio S. (2002) Are artificial reefs comparable to neighbouring natural rocky areas? A mollusc case study in the Gulf of Castellammare (NW Sicily). *ICES Journal of Marine Science*, 59, 127–131.*

Balash C., Sterling D. & Broadhurst M.K. (2016) Progressively evaluating a penaeid W trawl to improve eco-efficiency. *Fisheries Research*, 181, 148–154.*

Bamber R.N. (1990) Power station thermal effluents and marine crustaceans. *Journal of Thermal Biology*, 15, 91–96.

Bannister R.C.A., Addison J.T. & Lovewell S.R.J. (1994) Growth, movement, recapture rate and survival of hatchery-reared lobsters (*Homarus gammarus* (Linnaeus, 1758)) released into the wild on the English east coast. *Crustaceana*, 67, 156–172.*

Barnett P.R.O. (1972). Effects of warm water effluents from power stations on marine life. *Proceedings of the Royal Society of London: B,* 180, 497–509.

Barrio-Froján C.R., Cooper K.M., Bremner J., Defew E.C., Hussin W.M.W. & Paterson D.M. (2011) Assessing the recovery of functional diversity after sustained sediment screening at an aggregate dredging site in the North Sea. *Estuarine, Coastal and Shelf Science*, 92, 358–366.*

Barry S.C., Hayes K.R., Hewitt C.L., Behrens H.L., Dragsund E. & Bakke S.M. (2008) Ballast water risk assessment: principles, processes and methods. *ICES Journal of Marine Science*, 65, 121–131.

Batsleer J., Marchal P., Vaz S., Vermard V, Rijnsdorp A.D., Poos J.J. (2018) Exploring habitat credits to manage the benthic impact in a mixed fishery. *Marine Ecology Progress Series*, 586, 167–179.

Bax N., Williamson A., Aguero M., Gonzalez E. & Geeves W. (2003) Marine invasive alien species: a threat to global biodiversity. *Marine Policy*, 27, 313–323.

Bayraktarov E., Saunders M.I., Abdullah S., Mills M., Beher J., Possingham H.P., Mumby P.J. & Lovelock C.E. (2016) The cost and feasibility of marine coastal restoration. *Ecological Applications*, 26, 1055–1074.*

Benbow S., Humber F., Oliver T.A., Oleson K.L.L., Raberinar, D., Nado, M., Ratsimbazaf, H. & Harris A. (2014) Lessons learnt from experimental temporary octopus fishing closures in south-west Madagascar: benefits of concurrent closures. *African Journal of Marine Science*, 36, 31–37.*

Béné C. & Tewfik A. (2003) Biological evaluation of marine protected area: evidence of crowding effect on a protected population of queen conch in the Caribbean. *Marine Ecology*, 24, 45–58.*

Bergman M.J.N. & Van Santbrink J.W. (2000) Mortality in megafaunal benthic populations caused by trawl fisheries on the Dutch continental shelf in the North Sea in 1994. *ICES Journal of Marine Science,* 57, 1321–1331.*

Bergman M.J.N., Ubels S.M., Duineveld G.C.A. & Meesters E.W.G. (2015) Effects of a 5-year trawling ban on the local benthic community in a wind farm in the Dutch coastal zone. *ICES Journal of Marine Science*, 72, 962–972.*

Bernhardt J.R. & Leslie H.M. (2013) Resilience to climate change in coastal marine ecosystems. *Annual Review of Marine Science,* 5, 371–392.

Beukers-Stewart B.D., Vause B.J., Mosley M.W.J., Rossetti H.L. & Brand A.R. (2005). Benefits of closed area protection for a population of scallops. *Marine Ecology Progress Series,* 298, 189–204.*

Bilkovic D.M., Havens K.J., Stanhope D.M. & Angstadt K.T. (2012) Use of fully biodegradable panels to reduce derelict pot threats to marine fauna. *Conservation Biology,* 26, 957–966.

Birchenough S.N. & Frid C.L. (2009) Macrobenthic succession following the cessation of sewage sludge disposal. *Journal of Sea Research*, 62, 258–267.*

Birchenough S.N., Boyd S.E., Vanstaen K., Coggan R.A. & Limpenny D.S. (2010) Mapping an aggregate extraction site off the Eastern English Channel: a methodology in support of monitoring and management. *Estuarine, Coastal and Shelf Science*, 87, 420–430.*

Bishop M., Peterson C.H., Summerson H.C. & Gaskill D. (2005) Effects of harvesting methods on sustainability of a bay scallop fishery: dredging uproots seagrass and displaces recruits. Fishery Bulletin, 103, 712–719.

Bishop M.J., Krassoi F.R., McPherson R.G., Brown K.R., Summerhayes S.A., Wilkie E.M. & O'Connor W.A. (2010) Change in wild-oyster assemblages of Port Stephens, NSW, Australia, since commencement of non-native Pacific oyster (*Crassostrea gigas*) aquaculture. *Marine and Freshwater Research*, 61, 714–723.

Blyth R.E., Kaiser M. J., Edwards-Jones G. & Hart P.J.B. (2002) Voluntary management in an inshore fishery has conservation benefits. *Environmental Conservation*, 29, 493–508.

Blyth R.E., Kaiser M.J., Edwards-Jones G. & Hart P.J.B. (2004) Implications of a zoned fishery management system for marine benthic communities. *Journal of Applied Ecology*, 41, 951–961.*

Boehlert G. & Gill A. (2010) Environmental and ecological effects of ocean renewable energy development: A current synthesis. *Oceanography*, 23, 68–81.

Bohnsack J.A. & Sutherland D.L. (1985) Artificial reef research: a review with recommendations for future priorities. *Bulletin of Marine Science*, 37, 11–39.

Bonne W.M. (2010) Macrobenthos characteristics and distribution, following intensive sand extraction from a subtidal sandbank. *Journal of Coastal Research*, 141–150.*

Boulcott P., Millar C.P. & Fryer R.J. (2014) Impact of scallop dredging on benthic epifauna in a mixed-substrate habitat. *ICES Journal of Marine Science,* 71, 834–844.*

Bourque A.S. & Fourqurean J.W. (2014) Effects of common seagrass restoration methods on ecosystem structure in subtropical seagrass meadows. *Marine Environmental Research*, 97, 67–78.*

Boyd S.E. & Rees H.L. (2003) An examination of the spatial scale of impact on the marine benthos arising from marine aggregate extraction in the central English Channel. *Estuarine, Coastal and Shelf Science*, 57, 1–16.*

Brčić J., Herrmann B. & Sala A. (2017) Can a square-mesh panel inserted in front of the cod end improve size and species selectivity in Mediterranean trawl fisheries? *Canadian Journal of Fisheries and Aquatic Sciences*, 75, 704–713.

Brčić J., Herrmann B., De Carlo F. & Sala A. (2015) Selective characteristics of a shark-excluding grid device in a Mediterranean trawl. *Fisheries Research*, 172, 352–360.

Breitburg D., Levin L.A., Oschlies A., Grégoire M., Chavez F.P., Conley D.J., Garçon V., Gilbert D., Gutiérrez D., Isensee K., Jacinto G.S., Limburg K.E., Montes I., Naqvi S.W.A., Pitcher G.C., Rabalais N.N., Roman M.R., Rose K.A., Seibel B.A., Telszewski M., Yasuhara M. & Zhang J. (2018) Declining oxygen in the global ocean and coastal waters. *Science,* 359.

Brenner M., Fraser D., Van Nieuwenhove K., O'Beirn F., Buck B.H., Mazurié J., Thorarinsdottir G., Dolmer P., Sanchez-Mata A., Strand O. & Flimlin G. (2014) Bivalve aquaculture transfers in Atlantic Europe. Part B: environmental impacts of transfer activities. *Ocean & Coastal Management*, 89, 139–146.

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. *

Brigitte S., Fowler A.M., Macreadie P.I., Palandro D.A., Aziz A.C. & Booth D.J. (2018) Decommissioning of offshore oil and gas structures–Environmental opportunities and challenges. *Science of the Total Environment*, 658, 973–981.

Briones-Fourzàn P. & Lozano-Álvarez E. (2001) Effects of artificial shelters (Casitas) on the abundance and biomass of juvenile spiny lobsters *Panulirus argus* in a habitat-limited tropical reef lagoon. *Marine Ecology Progress Series,* 221, 221–232.*

Brix H. (1994) Use of constructed wetlands in water pollution control: historical development, present status, and future perspectives. *Water Science and Technology,* 30, 209–223.

Broadhurst M.K., Butcher P.A. & Cullis B.R. (2014) Effects of mesh size and escape gaps on discarding in an Australian giant mud crab (*Scylla serrata*) trap fishery. *PloS One*, 9, e106414.*

Broadhurst M.K., Millar R.B. & Brand C.P. (2010) Diamond-vs. square-mesh codend selectivity in southeastern Australian estuarine squid trawls. *Fisheries Research*, 102, 276–285.

Broadhurst M.K., Sterling D.J. & Millar R.B. (2013a) Progressing more environmentally benign penaeid-trawling systems by comparing Australian single- and multi-net configurations. *Fisheries Research,* 146, 7–17.

Broadhurst M.K., Sterling D.J. & Millar R.B. (2013b) Relative engineering and catching performances of paired penaeid-trawling systems. *Fisheries Research,* 143, 143–152.

Brooks T.M., Mittermeier R.A., Mittermeier C.G., Da Fonseca G.A., Rylands A.B., Konstant W.R., Flick P., Pilgrim J., Oldfield S., Magin G. & Hilton-Taylor C. (2002) Habitat loss and extinction in the hotspots of biodiversity. *Conservation Biology*, 16, 909–923.

Brown L.A., Furlong J.N., Brown K.M. & La Peyre M.K. (2014) Oyster reef restoration in the northern Gulf of Mexico: effect of artificial substrate and age on nekton and benthic macroinvertebrate assemblage use. *Restoration Ecology*, 22, 214–222.*

Brumbaugh R.D. & Coen L.D. (2009) Contemporary approaches for small-scale oyster reef restoration to address substrate versus recruitment limitation: a review and comments relevant for the Olympia oyster, *Ostrea lurida* Carpenter 1864. *Journal of Shellfish Research*, 28, 147–161.

Bryan G.W., Burt G.R., Gibbs P.E. & Pascoe P.L. (1993) *Nassarius reticulatus* (Nassariidae: Gastropoda) as an indicator of tributyltin pollution before and after TBT restrictions. *Journal of the Marine Biological Association of the United Kingdom*, 73, 913–929.*

Bull M.F. (1989) *The New Zealand scallop fishery: a brief review of the fishery and its management*. Edited by: MLC Dredge, WF Zacharin and LM Joli, 42.

Bulleri F. & Airoldi L. (2005) Artificial marine structures facilitate the spread of a non-indigenous green alga, *Codium fragile* ssp. *tomentosoides*, in the north Adriatic Sea. *Journal of Applied Ecology*, 42, 1063–1072.

Burgos-León A., Pérez-Castañeda R. & Defeo O. (2009) Discards from the artisanal shrimp fishery in a tropical coastal lagoon of Mexico: spatio-temporal patterns and fishing gear effects. *Fisheries Management and Ecology*, 16, 130–138.*

Burridge C.Y., Pitcher C.R., Hill B.J., Wassenberg T.J. & Poiner I.R. (2006) A comparison of demersal communities in an area closed to trawling with those in adjacent areas open to trawling: a study in the Great Barrier Reef Marine Park, Australia. *Fisheries Research*, 79, 64–74.*

Burridge L., Weis J.S., Cabello F., Pizarro J. & Bostick K. (2010) Chemical use in salmon aquaculture: a review of current practices and possible environmental effects. *Aquaculture*, 306, 7–23.

Burton, W.H., Farrar, J.S., Steimle, F., & Conlin, B. (2002) Assessment of out-of-kind mitigation success of an artificial reef deployed in Delaware Bay, USA. *ICES Journal of Marine Science*, 59, 106–110.*

Bustamante M., Bevilacqua S., Tajadura J., Terlizzi A. & Saiz-Salinas J.I. (2012) Detecting human mitigation intervention: Effects of sewage treatment upgrade on rocky macrofaunal assemblages. *Marine Environmental Research,* 80, 27–37.*

Butt N. (2007) The impact of cruise ship generated waste on home ports and ports of call: A study of Southampton. *Marine Policy*, 31, 591–598.

Cabello F.C. (2006) Heavy use of prophylactic antibiotics in aquaculture: a growing problem for human and animal health and for the environment. *Environmental Microbiology,* 8, 1137–1144

Caddy J.F., Defeo D. & Defeo O. (2003) *Enhancing or Restoring the Productivity of Natural Populations of Shellfish and Other Marine Invertebrate Resources* (Vol. 448). Food & Agriculture Organisation.

Calderwood J., O'Connor N.E. & Roberts D. (2015) Effects of baited crab pots on cultivated mussel (*Mytilus edulis*) survival rates. *ICES Journal of Marine Science*, 72, 1802–1810.

Campbell M.L. & Hewitt C.L. (1999) Vectors, shipping and trade. Pages 45–60 in: Hewitt C L, Campbell ML, Thresher RE, Martin RB (eds.). *The Introduced Species of Port Phillip Bay, Victoria*. Centre for Research on Introduced Marine Pests, CSIRO Marine Research, Hobart.

Campbell M.L. & Hewitt C.L. (2008) Introduced marine species risk assessment – aquaculture. Pages 121–133 in: M.G. Bondad-Reantaso; J.R. Arthur. & R.P. Subasinghe (eds). *Understanding and applying risk analysis in aquaculture*. FAO Fisheries and Aquaculture Technical Paper. No. 519. Rome, FAO.

Campbell M.L. (2009). An overview of risk assessment in a marine biosecurity context. Chapter 20. 99 353–373 in: Rilov G & Crooks J (eds.). *Biological Invasions in Marine Ecosystems. Ecological, Management, and Geographic Perspectives*. Heidelberg, Germany: Springer

Campbell M.L. (2011) Assessing biosecurity risk associated with the importation of microalgae. *Environmental Research,* 111, 989–998.

Campbell M.L., Grage A., Mabin C. & Hewitt C.L. (2009) Conflict between International Treaties: Failing to mitigate the effects of introduced marine species. *Dialogue,* 28, 46–56

Campbell M.L., King S., Heppenstall L.D., van Gool E., Martin R. & Hewitt C.L. (2017) Aquaculture and urban marine structures facilitate native and non-indigenous species transfer through generation and accumulation of marine debris. *Marine Pollution Bulletin*, 123, 304–312.

Cariglia, N., Wilson, S.K., Graham, N.A.J., Fisher, R., Robinson, J., Aumeeruddy, R., Quatre, R., & Polunin, N.V.C. (2013) Sea cucumbers in the Seychelles: effects of marine protected areas on high-value species. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 23, 418–428.*

Carter L., Burnett D., Drew S., Marle G., Hagadorn L., Bartlett McNeil D. & Irvine N. (2009) *Submarine cables and the oceans – connecting the world.* UNEP-WCMC Biodiversity Series No. 31. ICPC/UNEP/UNEP-WCMC.

Carvalho S., Moura A., Cúrdia J., da Fonseca L.C. & Santos M.N. (2013) How complementary are epibenthic assemblages in artificial and nearby natural rocky reefs? *Marine Environmental Research*, 92, 170–177.*

Castège I., Milon E., Fourneau G. & Tauzia A. (2016) First results of fauna community structure and dynamics on two artificial reefs in the south of the Bay of Biscay (France). *Estuarine, Coastal and Shelf Science*, 179, 172–180.*

Catchpole T.L., Revill A.S., Innes J. & Pascoe S. (2008) Evaluating the efficacy of technical measures: a case study of selection device legislation in the UK *Crangon crangon* (brown shrimp) fishery. *ICES Journal of Marine Science*, 65, 267–275.*

Ceccherelli G. Pinna S. & Sechi N. (2009) Evaluating the effects of protection on *Paracentrotus lividus* distribution in two contrasting habitats. *Estuarine, Coastal and Shelf Science*, 81, 59–64.*

Ceccherelli, G., Casu, D., Pala, D., Pinna, S., & Sechi, N. (2006) Evaluating the effects of protection on two benthic habitats at Tavolara-Punta Coda Cavallo MPA (North-East Sardinia, Italy). *Marine Environmental Research,* 61, 171–185.*

Chandler J., White D., Techera E. J., Gourvenec S. & Draper S. (2017) Engineering and legal considerations for decommissioning of offshore oil and gas infrastructure in Australia. *Ocean Engineering*, 131, 338–347.

Chávez-Crooker P. & Obreque-Contreras J. (2010) Bioremediation of aquaculture wastes. *Current opinion in Biotechnology*, 21, 313–317.

Cheung W.W., Lam V.W., Sarmiento J.L., Kearney K., Watson R. & Pauly D. (2009) Projecting global marine biodiversity impacts under climate change scenarios. *Fish and Fisheries*, 10, 235–251.

Chícharo L., Chícharo A., Gaspar M., Alves F. & Regala J. (2002) Ecological characterization of dredged and non-dredged bivalve fishing areas off south Portugal. *Journal of the Marine Biological Association of the United Kingdom*, 82, 41–50.*

Chou L.M. (2006) Marine Habitats in One of the World's Busiest Harbours. In: Wolanski E. (eds) *The Environment in Asia Pacific Harbours*. Springer, Dordrecht.

Chung I.K., Kang Y.H., Yarish C., Kraemer G.P. & Lee J.A. (2002) Application of seaweed cultivation to the bioremediation of nutrient-rich effluent. *Algae,* 17, 187–194.

Cigliano J.A., Meyer R., Ballard H.L., Freitag A., Phillips T.B. & Wasser A. (2015) Making marine and coastal citizen science matter. *Ocean & Coastal Management*, 115, 77–87.

Clark M.R. & Koslow J.A. (2007) Impacts of fisheries on seamounts. *Seamounts: Ecology, Fisheries, and Conservation*, 12, 413–441.

Clark R.B., Frid C. & Attrill M. (2001) *Marine pollution* (5th ed). Oxford University Press, Oxford; New York.

Clark S. & Edwards A.J. (1999) An evaluation of artificial reef structures as tools for marine habitat rehabilitation in the Maldives. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 9, 5–21.

Clarke Murray C., Pakhomov E.A. & Therriault T.W. (2011) Recreational boating: a large unregulated vector transporting marine invasive species. *Diversity and Distributions*, 17, 1161–1172.

Coates D.A., Deschutter Y., Vincx M. & Vanaverbeke J. (2014) Enrichment and shifts in macrobenthic assemblages in an offshore wind farm area in the Belgian part of the North Sea. *Marine Environmental Research*, 95, 1–12.

Coates D.A., Kapasakali D.A., Vincx M. & Vanaverbeke J. (2016) Short-term effects of fishery exclusion in offshore wind farms on macrofaunal communities in the Belgian part of the North Sea. *Fisheries Research*, 179, 131–138.*

Coates J.H., Hovel K.A., Butler J.L., Peter Klimley A. & Morgan S.G. (2013) Movement and home range of pink abalone *Haliotis corrugata*: Implications for restoration and population recovery. *Marine Ecology Progress Series*, 486, 189–201.*

Cole R.G. & Keuskamp D. (1998) Indirect effects of protection from exploitation: Patterns from populations of *Evechinus chloroticus* (Echinoidea) in northeastern New Zealand. *Marine Ecology Progress Series*, 173, 215–226.*

Coleman, M.A., Palmer-Brodie, A., & Kelaher, B.P. (2013) Conservation benefits of a network of marine reserves and partially protected areas. Biological conservation, 167, 257–264.*

Colleton N., Lakshman V., Flood K., Birnbaum M., Mcmillan K. & Lin A. (2016) Concepts and practice in the emerging use of games for marine education and conservation. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 26, 213–224.

Collie J.S., Hall S.J., Kaiser M.J. & Poiner I.R. (2000) A quantitative analysis of fishing impacts on shelf-sea benthos. *Journal of Animal Ecology*, 69, 785–798.

Collins K.J., Jensen A.C., Mallinson J.J., Roenelle V. & Smith I.P. (2002) Environmental impact assessment of a scrap tyre artificial reef. *ICES Journal of Marine Science*, 59, 243–249.*

Cooke S.J. & Cowx I.G. (2006) Contrasting recreational and commercial fishing: Searching for common issues to promote unified conservation of fisheries resources and aquatic environments. *Biological Conservation,* 128, 93–108.

Cooper K., Ware S., Vanstaen K. & Barry J. (2011) Gravel seeding - A suitable technique for restoring the seabed following marine aggregate dredging? *Estuarine, Coastal and Shelf Science,* 91, 121–132.*

Cooper K.M. (2013) Setting limits for acceptable change in sediment particle size composition: Testing a new approach to managing marine aggregate dredging. *Marine Pollution Bulletin,* 73, 86–97.

Cordes E.E., Jones D.O., Schlacher T.A., Amon D.J., Bernardino A.F., Brooke S., Carney R., DeLeo D.M., Dunlop K.M., Escobar-Briones E.G. & Gates A.R. (2016) Environmental impacts of the deep-water oil and gas industry: a review to guide management strategies. *Frontiers in Environmental Science*, 4, 58.

Costello C., Gaines S.D. & Lynham J. (2008) Can catch shares prevent fisheries collapse? *Science*, 321, 1678–1681

Cotton S. & Wedekind C. (2007) Control of introduced species using Trojan sex chromosomes. *Trends in Ecology & Evolution*, 22, 441–443.

Cox C. & Hunt J.H. (2005) Change in size and abundance of Caribbean spiny lobsters *Panulirus argus* in a marine reserve in the Florida Keys National Marine Sanctuary, USA. *Marine Ecology Progress Series*, 294, 227–239.*

Coyer J.A., Ambrose R.F., Engle J.M. & Carroll J.C. (1993) Interactions between corals and algae on a temperate zone rocky reef: mediation by sea urchins. *Journal of Experimental Marine Biology and Ecology,* 167, 21–37.

Crisp J.A., Loneragan N.R., Tweedley J.R., D'Souza F.M.L. & Poh B. (2018) Environmental factors influencing the reproduction of an estuarine penaeid population and implications for management. *Fisheries Management and Ecology*, 25, 203–219.*

Crochelet E., Roberts J., Lagabrielle E., Obura D., Petit M. & Chabanet P. (2016) A model-based assessment of reef larvae dispersal in the Western Indian Ocean reveals regional connectivity patterns—Potential implications for conservation policies. *Regional Studies in Marine Science*, 7, 159–167.

Currie D.R. & Parry G.D. (1996) Effects of scallop dredging on a soft sediment community: a large-scale experimental study. *Marine Ecology Progress Series*, 134, 131–150.

Curtis J.M., Ribeiro J., Erzini K. & Vincent A.C. (2007) A conservation trade-off? Interspecific differences in seahorse responses to experimental changes in fishing effort. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 17, 468–484.*

Dafforn K.A., Mayer-Pinto M., Morris R.L. & Waltham N.J. (2015) Application of management tools to integrate ecological principles with the design of marine infrastructure. *Journal of Environmental Management,* 158, 61–73.

Dannheim J., Brey T., Schröder A., Mintenbeck K., Knust R. & Arntz W.E. (2014) Trophic look at soft-bottom communities — Short-term effects of trawling cessation on benthos. *Journal of Sea Research,* 85, 18–28.*

Danovaro R., Gambi C., Mazzola A. & Mirto S. (2002) Influence of artificial reefs on the surrounding infauna: analysis of meiofauna. *ICES Journal of Marine Science*, 59, 356–362.*

Daudi L.N., Uku J.N. & De Troch M. (2013) Role of the source community for the recovery of seagrass associated meiofauna: a field colonisation experiment with seagrass mimics in Diani Beach, Kenya. *African Journal of Marine Science*, 35, 1–8.*

Davidson R.J., Villouta E., Cole R.G. & Barrier R.G.F. (2002) Effects of marine reserve protection on spiny lobster (*Jasus edwardsii*) abundance and size at Tonga Island Marine Reserve, New Zealand. *Aquatic Conservation: Marine and Freshwater Ecosystems,* 12, 213–227.*

Davies R.W.D., Cripps S.J., Nickson A. & Porter G. (2009) Defining and estimating global marine fisheries unwanted catch. *Marine Policy*, 33, 661–672.

Davies T.E., Maxwell S.M., Kaschner K., Garilao C. & Ban N.C. (2017) Large marine protected areas represent biodiversity now and under climate change. *Scientific Reports*, 7, 9569.

Davies T.W., Coleman M., Griffith K.M. & Jenkins S.R. (2015) Night-time lighting alters the composition of marine epifaunal communities. *Biology Letters*, 11, 20150080.

Davies T.W., Duffy J.P., Bennie J. & Gaston K.J. (2014) The nature, extent, and ecological implications of marine light pollution. *Frontiers in Ecology and the Environment*, 12, 347–355.

Davies, C.E., Johnson, A.F., Wootton, E.C., Greenwood, S.J., Clark, K.F., Vogan, C.L., & Rowley, A.F. (2014) Effects of population density and body size on disease ecology of the European lobster in a temperate marine conservation zone. *ICES Journal of Marine Science*, 72, 128–138.*

De Groot S.J. (1982) The impact of laying and maintenance of offshore pipelines on the marine environment and the North Sea fisheries. *Ocean Management*, 8, 1–27.

De Groot S.J. (1996) The physical impact of marine aggregate extraction in the North Sea. *ICES Journal of Marine Science*, 53, 1051–1053.

de Jong M.F., Baptist M.J., Lindeboom H.J. & Hoekstra P. (2015) Short-term impact of deep sand extraction and ecosystem-based landscaping on macrozoobenthos and sediment characteristics. *Marine Pollution Bulletin,* 97, 294–308.

De Paula A.F. Fleury B.G. Lages B.G. & Creed J.C. (2017) Experimental evaluation of the effects of management of invasive corals on native communities. *Marine Ecology Progress Series,* 572, 141–154.*

de Soto N.A. (2016) Peer-reviewed studies on the effects of anthropogenic noise on marine invertebrates: from scallop larvae to giant squid. In *The Effects of Noise on Aquatic Life II* (17–26). Springer, New York, NY.

Deegan L.A., Wright A., Ayvazian S.G., Finn J.T., Golden H., Merson R.R. & Harrison J. (2002) Nitrogen loading alters seagrass ecosystem structure and support of higher trophic levels. A*quatic Conservation: Marine and Freshwater Ecosystems*, 12, 193–212.*

Defoirdt T., Sorgeloos P. & Bossier P. (2011) Alternatives to antibiotics for the control of bacterial disease in aquaculture. *Current opinion in microbiology*, 14, 251–258.

Dehon D.D. (2010) Investigating the use of bioengineered oyster reefs as a method of shoreline protection and carbon storage. Master's Thesis. Louisiana State University, 1084.

dela Cruz D.W., Villanueva R.D. & Baria M.V.B. (2014) Community-based, low-tech method of restoring a lost thicket of *Acropora* corals. *ICES Journal of Marine Science*, 71, 1866–1875.*

Delgado G.A. & Glazer R.A. (2007) Interactions between translocated and native queen conch *Strombus gigas*: evaluating a restoration strategy. *Endangered Species Research*, 3, 259–266.*

Demers M.C.A., Davis A.R. & Knott N.A. (2013) A comparison of the impact of 'seagrass-friendly'boat mooring systems on *Posidonia australis*. *Marine Environmental Research*, 83, 54–62.

Depestele J., Degrendele K., Esmaeili M., Ivanović A., Kröger S., O'neill F.G., Parker R., Polet H., Roche M., Teal, L.R. & Vanelslander B. (2018) Comparison of mechanical disturbance in soft sediments due to tickler-chain SumWing trawl vs. electro-fitted PulseWing trawl. *ICES Journal of Marine Science*, 76, 312–329.

Desilva S., Ingram B.A., Gooley G.F. & McKinon L.J. (2000) Aquaculture-agriculture systems integration: an Australian perspective. *Fisheries Management and Ecology*.

Desprez M. (2000) Physical and biological impact of marine aggregate extraction along the French coast of the Eastern English Channel: short-and long-term post-dredging restoration. I*CES Journal of Marine Science,* 57, 1428–1438.*

Devictor V., Whittaker R.J. & Beltrame C. (2010) Beyond scarcity: citizen science programmes as useful tools for conservation biogeography. *Diversity and distributions*, 16, 354–362.

Díaz D., Mallol S., Parma A.M. & Goñi R. (2016) A 25-year marine reserve as proxy for the unfished condition of an exploited species. *Biological Conservation*, 203, 97–107.*

Dobson A.P., Bradshaw A.D. & Baker A.Á. (1997) Hopes for the future: restoration ecology and conservation biology. *Science*, 277, 515–522.

Douvere F. (2008) The importance of marine spatial planning in advancing ecosystem-based sea use management. *Marine Policy*, 32, 762–771.

Duineveld G.C.A., Bergman M.J.N. & Lavaleye M.S.S. (2007) Effects of an area closed to fisheries on the composition of the benthic fauna in the southern North Sea. *ICES Journal of Marine Science,* 64, 899–908.*

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.

Easterling D.R., Meehl G.A., Parmesan C., Changnon S.A., Karl T.R. & Mearns L.O. (2000) Climate extremes: observations, modeling, and impacts. Science 289, 2068–2074.

Edgar G.J., Stuart-Smith R.D., Willis T.J., Kininmonth S., Baker S.C., Banks S., Barrett N.S., Becerro M.A., Bernard A.T., Berkhout J. & Buxton C.D. (2014) Global conservation outcomes depend on marine protected areas with five key features. *Nature*, 506, 216–220.

Ekins P., Vanner R. & Firebrace J. (2006) Decommissioning of offshore oil and gas facilities: A comparative assessment of different scenarios. *Journal of Environmental Management*, 79, 420–438.

Epstein G. & Smale D.A. (2017) *Undaria pinnatifida*: A case study to highlight challenges in marine invasion ecology and management. *Ecology and Evolution*, 7, 8624–8642.

Erzini K., Gonçalves J.M., Bentes L., Moutopoulos D.K., Casal J.A.H., Soriguer M.C., Puente E., Errazkin L.A. & Stergiou K.I. (2006) Size selectivity of trammel nets in southern European small-scale fisheries. *Fisheries Research*, 79, 183–201.

Esteves F.A., Caliman A., Santangelo J.M., Guariento R.D., Farjalla V.F. & Bozelli R.L. (2008) Neotropical coastal lagoons: an appraisal of their biodiversity, functioning, threats and conservation management. *Brazilian Journal of Biology*, 68, 967–981.

Fabi G., Luccarini F., Panfili M., Solustri C. & Spagnolo A. (2002) Effects of an artificial reef on the surrounding soft-bottom community (central Adriatic Sea). *ICES Journal of Marine Science*, 59, 343–349.*

Falace A., Tamburello L., Guarnieri G., Kaleb, S., Papa L. & Fraschetti S. (2018) Effects of a glyphosate-based herbicide on Fucus virsoides (Fucales, Ochrophyta) photosynthetic efficiency. *Environmental Pollution*, 243, 912–918.

Fariñas-Franco J.M. & Roberts D. (2014) Early faunal successional patterns in artificial reefs used for restoration of impacted biogenic habitats. *Hydrobiologia*, 727,75–94.*

Fariñas-Franco J.M., Allcock L., Smyth D. & Roberts D. (2013) Community convergence and recruitment of keystone species as performance indicators of artificial reefs. *Journal of Sea Research,* 78, 59–74.*

Field R. & Struzeski Jr. E. J. (1972) Management and control of combined sewer overflows. *Journal (Water Pollution Control Federation)*, 1393–1415.

Firth L.B., Thompson R.C., Bohn K., Abbiati M., Airoldi L., Bouma T.J., Bozzeda F., Ceccherelli V.U., Colangelo M.A., Evans A., Ferrario F., Hanley M.E., Hinz H., Hoggart S.P.G., Jackson J.E., Moore P., Morgan E.H., Perkol-Finkel S., Skov M.W., Strain E.M., van Belzen J. & Hawkins S.J. (2014) Between a rock and a hard place: Environmental and engineering considerations when designing coastal defence structures. *Coastal Engineering*, 87, 122–135.

Fitridge I., Dempster T., Guenther J. & de Nys R., (2012) The impact and control of biofouling in marine aquaculture: a review. *Biofouling*, 28, 649–669.

Foden J., Rogers S.I. & Jones A.P. (2009) Recovery rates of UK seabed habitats after cessation of aggregate extraction. *Marine Ecology Progress Series*, 390, 15–26.*

Follesa M.C., Cannas R., Cau A., Cuccu D., Gastoni A., Ortu A., Pedoni C., Porcu C. & Cau A. (2011) Spillover effects of a Mediterranean marine protected area on the European spiny lobster *Palinurus elephas* (Fabricius, 1787) resource. *Aquatic Conservation: Marine and Freshwater Ecosystems,* 21, 564–572.

Fonteyne R. & Polet H. (2002) Reducing the benthos by-catch in flatfish beam trawling by means of technical modifications. *Fisheries Research*, 55, 219–230.*

Frandsen R.P., Eigaard O.R., Poulsen L.K., Tørring D., Stage B., Lisbjerg D. & Dolmer P. (2015) Reducing the impact of blue mussel (*Mytilus edulis*) dredging on the ecosystem in shallow water soft bottom areas. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 25, 162–173

Franke J.M. (2007) *The invasive species cookbook: conservation through gastronomy.* Bradford street Press, Wauwatosa , WI.

Fraschetti S., Terlizzi A., Bussotti S., Guarnieri G., D'Ambrosio P. & Boero F. (2005) Conservation of Mediterranean seascapes: analyses of existing protection schemes. *Marine Environmental Research*, 59, 309–332.

Fraschetti S., Terlizzi A., Micheli F., Benedetti-Cecchi L. & Boero F. (2002) Marine protected areas in the Mediterranean Sea: objectives, effectiveness and monitoring. *Marine Ecology*, 23, 190–200.

Frumkes D.R. (2002) The status of the California Rigs-to-Reefs Programme and the need to limit consumptive fishing activities. *ICES Journal of Marine Science*, 59, 272–276.

Fuentes J., Molares J. & Villalba A. (1998) Growth, mortality and parasitization of mussels cultivated in the Ría de Arousa (NW Spain) from two sources of seed: intertidal rocky shore vs. collector ropes. Aquaculture, 162, 231–240.

Fukunaga A. & Bailey-Brock J.H. (2008) Benthic infaunal communities around two artificial reefs in Mamala Bay, Oahu, Hawaii. *Marine Environmental Research*, 65, 250–263.*

Furevik D.M., Humborstad O.B., Jørgensen T. & Løkkeborg S. (2008) Floated fish pot eliminates bycatch of red king crab and maintains target catch of cod. *Fisheries Research*, 92, 23–27.*

Gabel F., Garcia X.F., Schnauder I. & Pusch M.T. (2012) Effects of ship-induced waves on littoral benthic invertebrates. *Freshwater Biology*, 57, 2425–2435.

Gabric A.J. & Bell P.R.F. (1993) Review of the effects of non-point nutrient loading on coastal ecosystems. *Marine and Freshwater Research,* 44, 261–283.

Gall S.C. & Thompson R.C. (2015) The impact of debris on marine life. *Marine Pollution Bulletin*, 92, 170–179.

Gallardi D. (2014) Effects of bivalve aquaculture on the environment and their possible mitigation: a review. *Fisheries and Aquaculture Journal*, 5, 1.

Game E.T., Lipsett-Moore G., Saxon E., Peterson N. & Sheppard S. (2011) Incorporating climate change adaptation into national conservation assessments. *Global Change Biology*, 17, 3150–3160.

Ganning B., Broman D. & Lindblad C. (1983) Uptake of petroleum hydrocarbons by the blue mussel (*Mytilus edulis* L.) after experimental oiling and high pressure, hot water shore cleaning. *Marine Environmental Research*, 10, 245–254.*

Gao Q.-F., Shin P.K.S., Xu W.Z. & Cheung S.G. (2008). Amelioration of marine farming impact on the benthic environment using artificial reefs as biofilters. *Marine Pollution Bulletin*. 57, 652–661.

Gaspar M.B., Dias M.D., Campos A., Monteiro C.C., Santos M.N., Chicharo A. & Chicharo L. (2001) The influence of dredge design on the catch of *Callista chione* (Linnaeus, 1758). *Hydrobiologia,* 465, 153–167.*

Gaspar M.B., Leitão F., Santos M.N., Chícharo L., Dias, M.D., Chícharo, A., & Monteiro C.C. (2003) A comparison of direct macrofaunal mortality using three types of clam dredges. *ICES Journal of Marine Science,* 60, 733–742.*

Gaspar M.B., Leitão F., Santos M.N., Sobral M., Chícharo L., Chícharo A. & Monteiro C.C. (2002) Influence of mesh size and tooth spacing on the proportion of damaged organisms in the catches of the Portuguese clam dredge fishery. *ICES Journal of Marine Science,* 59, 1228–1236.*

Gaston K.J., Davies T.W., Bennie J. & Hopkins J. (2012) Reducing the ecological consequences of night-time light pollution: options and developments. *Journal of Applied Ecology*, 49, 1256–1266.

Gelcich S. & Donlan C.J. (2015) Incentivizing biodiversity conservation in artisanal fishing communities through territorial user rights and business model innovation. *Conservation Biology*, 29, 1076–1085.

George L.M., De Santiago K., Palmer T.A. & Pollack J.B. (2015) Oyster reef restoration: effect of alternative substrates on oyster recruitment and nekton habitat use. *Journal of Coastal Conservation*, 19, 13–22.*

Ghosh A.K., Pattnaik A.K. & Ballatore T.J. (2006) Chilika Lagoon: Restoring ecological balance and livelihoods through re-salinization. *Lakes & Reservoirs: Research & Management*, 11, 239–255.

Gibbs P.E., Bryan G.W., Pascoe P.L. & Burt G.R. (2009) The use of the dog-whelk, *Nucella lapillus*, as an indicator of tributyltin (TBT) contamination. *Journal of the Marine Biological Association of the United Kingdom*, 67, 507.

Gil Fernández C., Paulo D., Serrão E.A. & Engelen A.H. (2016) Limited differences in fish and benthic communities and possible cascading effects inside and outside a protected marine area in Sagres (SW Portugal). *Marine Environmental Research,* 114, 12–23.*

Gilardi K.V., Carlson-Bremer D., June J.A., Antonelis K., Broadhurst G.,& Cowan T. (2010) Marine species mortality in derelict fishing nets in Puget Sound, WA and the cost/benefits of derelict net removal. *Marine Pollution Bulletin*, 60, 376–382.

Gilkinson K.D., Fader G.B.J., Gordon Jr D.C., Charron R., McKeown D., Roddick D., Kenchington E.L.R., MacIsaac K., Bourbonnais C., Vass P. & Liu Q. (2003) Immediate and longer-term impacts of hydraulic clam dredging on an offshore sandy seabed: effects on physical habitat and processes of recovery. *Continental Shelf Research*, 23, 1315–1336.

Gill A.B. (2005) Offshore renewable energy: ecological implications of generating electricity in the coastal zone. *Journal of Applied Ecology*, 42, 605–615.

Gimpel A., Stelzenmüller V., Grote B., Buck B.H., Floeter J., Núñez-Riboni I., Pogoda B. & Temming A. (2015) A GIS modelling framework to evaluate marine spatial planning scenarios: Co-location of offshore wind farms and aquaculture in the German EEZ. *Marine Policy,* 55, 102–115.

Gleason M., Feller E.M., Merrifield M., Copps S., Fujita R.O.D., Bell M., Rienecke S. & Cook C. (2013) A transactional and collaborative approach to reducing effects of bottom trawling. *Conservation Biology,* 27, 470–479.

Glen D. (2010) Modelling the impact of double hull technology on oil spill numbers. *Maritime Policy & Management*, 37, 475–487.

Gonçalves J.M.S., Bentes L., Coelho R., Monteiro P., Ribeiro J., Correia C., Lino P.G. & Erzini K. (2008) Non-commercial invertebrate discards in an experimental trammel net fishery. *Fisheries Management and Ecology*, 15, 199–1210.

Gonçalves J.M.S., Stergiou K.I., Hernando J.A., Puente E., Moutopoulos D.K., Arregi L., Soriguer M.C., Vilas C., Coelho R. & Erzini K. (2007) Discards from experimental trammel nets in southern European small-scale fisheries. *Fisheries Research*, 88, 5–14.*

Gorman D. & Dixon C. (2015) Reducing discards in a temperate prawn trawl fishery: a collaborative approach to bycatch research in South Australia. *ICES Journal of Marine Science*, 72, 2609–2617.*

Götz A., Kerwath S.E., Attwood C.G. & Sauer W.H.H. (2009) Effects of fishing on a temperate reef community in South Africa 2: Benthic invertebrates and algae. *African Journal of Marine Science*, 31, 253–262.*

Goudey C.A., Loverich G., Kite-Powell H. & Costa-Pierce B.A. (2001) Mitigating the environmental effects of mariculture through single-point moorings (SPMs) and drifting cages. *ICES Journal of Marine Science,* 58, 497–503.

Graham P.M., Palmer T.A. & Beseres Pollack J. (2017) Oyster reef restoration: substrate suitability may depend on specific restoration goals. *Restoration Ecology*, 25, 459–470.*

Green A.L., Fernandes L., Almany G., Abesamis R., McLeod E., Aliño P.M., White A.T., Salm R., Tanzer J. & Pressey R.L. (2014) Designing marine reserves for fisheries management, biodiversity conservation, and climate change adaptation. *Coastal Management*, 42, 143–159.

Green B.S. & Gardner C. (2009) Surviving a sea-change: survival of southern rock lobster (*Jasus edwardsii*) translocated to a site of fast growth. *ICES Journal of Marine Science,* 66, 656–664.*

Griffiths C.A., Langmead O.A., Readman J.A.J. & Tillin H.M. (2017) *Anchoring and Mooring Impacts in English and Welsh Marine Protected Areas: Reviewing sensitivity, activity, risk and management.* A report to Defra Impacts Evidence Group.

Grilo T.F., Cardoso P.G., Dolbeth M., Bordalo M.D. & Pardal M.A. (2011) Effects of extreme climate events on the macrobenthic communities' structure and functioning of a temperate estuary. Marine Pollution Bulletin, 62, 303–311.

Guest J.R., Todd P.A., Goh E., Sivalonganathan B.S. & Reddy K.P. (2008) Can giant clam (*Tridacna squamosa*) populations be restored on Singapore's heavily impacted coral reefs? *Aquatic Conservation: Marine and Freshwater Ecosyst*ems, 18, 570–579.*

Gumarov S.M., Shokanov T.A., Simmons S., Anokhin V.V., Benelkadi S. & Ji L. (2014) *Drill cuttings re-injection well design and completion: Best practices and lessons learned*. Society of Petroleum Engineers.

Gutzler B.C. Butler M.J. & Behringer D.C. (2015) Casitas: A location-dependent ecological trap for juvenile Caribbean spiny lobsters, *Panulirus argus*. *ICES Journal of Marine Science,* 72, 177–184.*

Hall-Spencer J., White N., Gillespie E., Gillham K. & Foggo A. (2006) Impact of fish farms on maerl beds in strongly tidal areas. *Marine Ecology Progress Series*, 326, 1–9.

Hammerstrom K.K., Kenworthy W.J., Whitfield P.E. & Merello M.F. (2007) Response and recovery dynamics of seagrasses *Thalassia testudinum* and *Syringodium filiforme* and macroalgae in experimental motor vessel disturbances. *Marine Ecology Progress Series* 345, 83–92.

Handley S.J., Willis T.J., Cole R.G., Bradley A., Cairney D.J., Brown S.N. & Carter M.E. (2014) The importance of benchmarking habitat structure and composition for understanding the extent of fishing impacts in soft sediment ecosystems. *Journal of Sea Research*, 86. 58–68.*

Hannah R.W., Lomeli M.J. & Jones S.A. (2013) Direct estimation of disturbance rates of benthic macroinvertebrates from contact with standard and modified ocean shrimp (*Pandalus jordani*) trawl footropes. *Journal of Shellfish Research*, 32, 551–558.

Hansen S.C. & Gosselin L.A. (2013) Do predators, handling stress or field acclimation periods influence the survivorship of hatchery-reared abalone *Haliotis kamtschatkana* outplanted into natural habitats? *Aquatic Conservation: Marine and Freshwater Ecosystems,* 23, 246–253.*

Hardiman N. & Burgin S. (2010) Recreational impacts on the fauna of Australian coastal marine ecosystems. *Journal of Environmental Management*, 91, 2096–2108.

Harley C.D., Randall Hughes A., Hultgren K.M., Miner B.G., Sorte C.J., Thornber C.S., Rodriguez L.F., Tomanek L. & Williams S.L. (2006) The impacts of climate change in coastal marine systems. *Ecology Letters*, 9, 228–241.

Harriott V.J., Davis D. & Banks S.A. (1997) Recreational diving and its impact in marine protected areas in eastern Australia. *Ambio*, 173–179.

Harris C.M., Thomas L., Falcone E.A., Hildebrand J., Houser D., Kvadsheim P.H., Lam F.P.A., Miller P.J., Moretti D.J., Read A.J. & Slabbekoorn H. (2018) Marine mammals and sonar: Dose-response studies, the risk-disturbance hypothesis and the role of exposure context. *Journal of Applied Ecology*, 55, 396–404.

Hartwick E.B., Wu R.S.S. & Parker D.B. (1982) Effects of a crude oil and an oil dispersant (Corexit 9527) on populations of the littleneck clam (*Protothaca staminea*). *Marine Environmental Research*, 6, 291–306.

He P. & Winger P.D. (2010) Effect of Trawling on the Seabed and Mitigation Measures to Reduce Impact. Pages 295–314 in: P. He (Ed.) *Behavior of Marine Fishes.*

Henry L.A., Harries D., Kingston P. & Roberts J.M. (2017) Historic scale and persistence of drill cuttings impacts on North Sea benthos. *Marine Environmental Research,* 129, 219–228.

Herbert R.J., Crowe T.P., Bray S. & Sheader M. (2009) Disturbance of intertidal soft sediment assemblages caused by swinging boat moorings. Hydrobiologia, 625, 105–116.

Herbert R.J., Humphreys J., Davies C.J., Roberts C., Fletcher S. & Crowe T.P. (2016) Ecological impacts of non-native Pacific oysters (*Crassostrea gigas*) and management measures for protected areas in Europe. *Biodiversity and Conservation*, 25, 2835–2865.

Hermsen J.M., Collie J.S. & Valentine P.C. (2003) Mobile fishing gear reduces benthic megafaunal production on Georges Bank. *Marine Ecology Progress Series,* 260, 97–108.*

Hewitt C.L. & Campbell M.L. (2007) Mechanisms for the prevention of marine bioinvasions for better biosecurity. *Marine Pollution Bulletin*, 55, 395–401.

Hewitt C.L. (2003). Marine biosecurity issues in the world oceans: global activities and Australian directions. *Ocean Yearbook*, 17, 193–212.

Hewitt C.L., Campbell M.L., McEnnulty F., Moore M.M., Murfet N.B., Robertson B. & Schaffelke B. (2005) Efficacy of physical removal of a marine pest: the introduced kelp *Undaria pinnatifida* in a Tasmanian Marine Reserve. *Biological Invasions*, 7, 251–263.

Hewitt C.L., Campbell M.L., Thresher R.E., Martin R.B., Boyd S., Cohen B.F., Currie D.R., Gomon M.F., Keough M.J., Lewis J.A., Lockett M.M., Mays N., McArthur M.A., O'Hara T.D., Poore G.C.B., Ross D.J., Storey M., Watson J.E. & Wilson R.S. (2004a) Introduced and cryptogenic species in Port Phillip Bay, Victoria, Australia*. Marine Biology*, 144, 183–202.

Hewitt C.L., Willing J., Bauckham Al., Cassidy A.M., Cox C.M.S., Jones L. & Wotton D.M. (2004b) New Zealand marine biosecurity: Delivering outcomes in a fluid environment*. New Zealand Journal of Marine and Freshwater Research*, 38, 429–438.

Hiddink J.G., Hutton T., Jennings S. & Kaiser M. J. (2006) Predicting the effects of area closures and fishing effort restrictions on the production, biomass, and species richness of benthic invertebrate communities. *ICES Journal of Marine Science*, 63, 822–830.

Hiddink J.G., Jennings S., Sciberras M., Szostek C.L., Hughes K.M., Ellis N., Rijnsdorp A.D., McConnaughey R.A., Mazor T., Hilborn R. & Collie J.S. (2017) Global analysis of depletion and recovery of seabed biota after bottom trawling disturbance. *Proceedings of the National Academy of Sciences,* 114, 8301–8306.

Hinz H. Tarrant D. Ridgeway A. Kaiser M.J. & Hiddink J.G. (2011) Effects of scallop dredging on temperate reef fauna. *Marine Ecology Progress Series*, 432, 91–102.*

Hinz H., Murray L.G., Malcolm F.R. & Kaiser M.J. (2012) The environmental impacts of three different queen scallop (*Aequipecten opercularis*) fishing gears. *Marine Environmental Research,* 73, 85–95.*

Hiscock K., Sharrock S., Highfield J. & Snelling D. (2010) Colonization of an artificial reef in south-west England ex-HMS Scylla. *Journal of the Marine Biological Association of the United Kingdom,* 90, 69–94.*

Hobday A.J., Tegner M.J. & Haaker P.L. (2000) Over-exploitation of a broadcast spawning marine invertebrate: decline of the white abalone. *Reviews in Fish Biology and Fisheries*, 10, 493–514.

Hoegh-Guldberg O. & Bruno J.F. (2010) The impact of climate change on the world's marine ecosystems. *Science*, 328, 1523–1528.

Hoffmann E. & Dolmer P. (2000) Effect of closed areas on distribution of fish and epibenthos. *ICES Journal of Marine Science*, 57, 1310–1314.*

Hollister C.D. & Nadis S. (1998) Burial of radioactive waste under the seabed. *Scientific American*, 278, 60–65.

Homziak J., Fonseca M.S. & Kenworthy W.J. (1982) Macrobenthic community structure in a transplanted eelgrass (*Zostera marina*) meadow. *Marine Ecology Progress Series*, 211–221.

Hoskin M.G., Coleman R.A., Von Carlshausen E. & Davis C.M. (2011) Variable population responses by large decapod crustaceans to the establishment of a temperate marine no-take zone. *Canadian Journal of Fisheries and Aquatic Sciences*, 68, 185–200.*

Howarth L.M., Dubois P., Gratton P., Judge M., Christie B., Waggitt J.J., Hawkins J.P., Roberts C.M. & Stewart B.D. (2017) Trade-offs in marine protection: Multispecies interactions within a community-led temperate marine reserve. *ICES Journal of Marine Science,* 74, 263–276.*

Howarth L.M., Pickup S.E., Evans L.E., Cross T.J., Hawkins J.P., Roberts C.M. & Stewart B.D. (2015) Sessile and mobile components of a benthic ecosystem display mixed trends within a temperate marine reserve. *Marine Environmental Research,* 107, 8–23.*

Huang H. W. & Chuang C. T. (2010). Fishing capacity management in Taiwan: Experiences and prospects. *Marine Policy*, 34, 70–76.

Hughes D.J., Poloczanska E.S. & Dodd J. (2008) Survivorship and tube growth of reef-building *Serpula vermicularis* (Polychaeta: Serpulidae) in two Scottish sea lochs. *Aquatic Conservation: Marine and Freshwater Ecosystems,* 18, 117–129.*

Huijbers C.M., Connolly R.M., Pitt K.A., Schoeman D.S., Schlacher T.A., Burfeind D.D., Steele C., Olds A.D., Maxwell P.S., Babcock R.C. & Rissik D. (2015) Conservation benefits of marine reserves are undiminished near coastal rivers and cities. *Conservation Letters*, 8, 312–319.*

Hulme P.E. (2009) Trade, transport and trouble: managing invasive species pathways in an era of globalization. *Journal of Applied Ecology*, 46, 10–18.

Hunter W.R. & Sayer M.D.J. (2009) The comparative effects of habitat complexity on faunal assemblages of northern temperate artificial and natural reefs. *ICES Journal of Marine Science*, 66, 691–698.*

Hutchison Z.L., Sigray P., He H., Gill A.B., King J. & Gibson C. (2018) *Electromagnetic Field (EMF) impacts on elasmobranch (shark, rays, and skates) and American lobster movement and migration from direct current cables*. Sterling (VA): US Department of the Interior, Bureau of Ocean Energy Management. OCS Study BOEM, 3.

Huvenne V.A.I., Bett B.J., Masson D.G., Le Bas,T.P. & Wheeler A.J. (2016) Effectiveness of a deep-sea cold-water coral Marine Protected Area, following eight years of fisheries closure. *Biological Conservation*, 200, 60–69.

Ido S. & Shimrit P.F. (2015) Blue is the new green–ecological enhancement of concrete based coastal and marine infrastructure. *Ecological Engineering*, 84, 260–272.

IMO Assembly Resolution 24/982 (2005) Revised guidelines for the identification and designation of Particularly Sensitive Sea Areas.

Inger R., Attrill M.J., Bearhop S., Broderick A.C., Grecian W.J., Hodgson D.J., Mills C., Sheehan E., Votier S.C., Witt M.J. & Godley B.J. (2009) Marine renewable energy: potential benefits to biodiversity? An urgent call for research. *Journal of Applied Ecology*, 46,1145–1153.

IPCC (2013) *Climate Change 2013: The Physical Science Basis*. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex & P.M. Midgley (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA..

Islam M.S. & Tanaka M. (2004) Impacts of pollution on coastal and marine ecosystems including coastal and marine fisheries and approach for management: a review and synthesis. *Marine pollution bulletin*, 48, 624–649.

Islam S. & Tanaka M. (2004) Impacts of pollution on coastal and marine ecosystems including coastal and marine fisheries and approach for management: a review and synthesis. *Marine Pollution Bulletin,* 48, 624–649.

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

Jack L. & Wing S.R. (2010) Maintenance of old-growth size structure and fecundity of the red rock lobster *Jasus edwardsii* among marine protected areas in Fiordland, New Zealand. *Marine Ecology Progress Series*, 404, 161–172.*

Jack L., Wing S.R. & McLeod R.J. (2009) Prey base shifts in red rock lobster *Jasus edwardsii* in response to habitat conversion in Fiordland marine reserves: implications for effective spatial management. *Marine Ecology Progress Series*, 381, 213–222.*

Jewett S.C., Feder H.M. & Blanchard A. (1999) Assessment of the benthic environment following offshore placer gold mining in the northeastern Bering Sea. *Marine Environmental Research*, 48, 91–122.*

Joaquim S., Gaspar M.B., Matias D., Ben-Hamadou R. & Arnold W.S. (2007) Rebuilding viable spawner patches of the overfished *Spisula solida* (Mollusca: Bivalvia): a preliminary contribution to fishery sustainability. *ICES Journal of Marine Science*, 65, 60–64.*

Johannessen P., Botnen H. & Tvedten Ø.F. (1994) Macrobenthos: before, during and after a fish farm. *Aquaculture Research*, 25, 55–66

Johnson C.R., Banks S.C., Barrett N.S., Cazassus F., Dunstan P.K., Edgar G.J., Frusher S.D., Gardner C., Haddon M., Helidoniotis F. & Hill K. L. (2011) Climate change cascades: Shifts in oceanography, species' ranges and subtidal marine community dynamics in eastern Tasmania. *Journal of Experimental Marine Biology and Ecology*, 400, 17–32.

Jones C.G., Lawton J.H. & Shachak M. (1994) Organisms as ecosystem engineers. Pages 130–147 in: *Ecosystem Management*. Springer, New York, NY.

Jones D.A. & Nithyanandan M. (2013) Recruitment of marine biota onto hard and soft artificially created subtidal habitats in Sabah Al-Ahmad Sea City, Kuwait. *Marine Pollution Bulletin*, 72, 351–356.*

Jones D.A., Ealey T., Baca B., Livesey S. & Al-Jamali F. (2007) Gulf desert developments encompassing a marine environment, a compensatory solution to the loss of coastal habitats by infill and reclamation: The case of the Pearl City Al-Khiran, Kuwait. *Aquatic Ecosystem Health & Management*, 10, 268–276.*

Jones D.A., Nithyanandan M. & Williams I. (2012) Sabah Al-Ahmad Sea City Kuwait: development of a sustainable manmade coastal ecsystem in a saline desert. *Aquatic Ecosystem Health Management*, 15, 82–90.

Kachel M.J. (2008) Particularly sensitive sea areas. Pages 1-184 in: *The IMO's Role in Protecting Vulnerable Marine Areas*. Berlin Heidelberg: Springer-Verlag.

Kaiser M.J. Spence F.E. & Hart P.J.B. (2000) Fishing-gear restrictions and conservation of benthic habitat complexity. *Conservation Biology*, 14, 1512–1525.*

Kaiser M.J., Clarke K.R., Hinz H., Austen M. C.V., Somerfield P.J. & Karakassis I. (2006) Global analysis of response and recovery of benthic biota to fishing. *Marine Ecology Progress Series*, 311, 1–14.

Kathman R.D., Brinkhurst R.O., Woods R.E. & Jeffries D.C. (1983) Benthic studies in Alice Arm and Hastings Arm, BC in relation to mine tailings dispersal. Institute of Ocean Sciences, Department of Fisheries and Oceans.

Katsanevakis S. (2016) Transplantation as a conservation action to protect the Mediterranean fan mussel *Pinna nobilis*. *Marine Ecology Progress Series*, 546, 113–122.

Katsanevakis S., Zenetos A., Belchior C. & Cardoso A.C. (2013) Invading European Seas: assessing pathways of introduction of marine aliens. *Ocean & Coastal Management*, 76, 64–74.*

Kelleher, G., & Kenchington, R.A. (1991) *Guidelines for establishing marine protected areas* (Vol. 3). IUCN.

Kennelly S.J. & Broadhurst M.K. (2014) Mitigating the bycatch of giant cuttlefish *Sepia apama* and blue swimmer crabs *Portunus armatus* in an Australian penaeid-trawl fishery. *Endangered Species Research*, 26, 161–166.*

Keppel G., Mokany K., Wardell-Johnson G.W., Phillips B.L., Welbergen J.A. & Reside A.E. (2015) The capacity of refugia for conservation planning under climate change. *Frontiers in Ecology and the Environment*, 13, 106–112.

Kilian J.V., Klauda R.J., Widman S., Kashiwagi M., Bourquin R., Weglein S. & Schuster J. (2012) An assessment of a bait industry and angler behavior as a vector of invasive species. *Biological Invasions,* 14, 1469–1481.

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.*

Kline E.R. & Stekoll M.S. (2001) Colonization of mine tailings by marine invertebrates. *Marine Environmental Research*, 51, 301–325.*

Königson S., Lövgren J., Hjelm J., Ovegård M., Ljunghager F. & Lunneryd S. G. (2015) Seal exclusion devices in cod pots prevent seal bycatch and affect their catchability of cod. *Fisheries Research*, 167, 114–122.

Koslow J.A., Gowlett-Holmes K., Lowry J.K., O'Hara T., Poore G.C.B. & Williams A. (2001) Seamount benthic macrofauna off southern Tasmania: Community structure and impacts of trawling. *Marine Ecology Progress Series,* 213, 111–125.*

Kraus C. & Carter L. (2018) Seabed recovery following protective burial of subsea cables - Observations from the continental margin. *Ocean Engineering,* 157, 251–261.

Kress N., Tom M. & Spanier E. (2002) The use of coal fly ash in concrete for marine artificial reefs in the southeastern Mediterranean: compressive strength, sessile biota, and chemical composition. *ICES Journal of Marine Science*, 59, 231–237.*

Krone R., Dederer G., Kanstinger P., Krämer P., Schneider C. & Schmalenbach I. (2017) Mobile demersal megafauna at common offshore wind turbine foundations in the German Bight (North Sea) two years after deployment-increased production rate of Cancer pagurus. *Marine Environmental Research*, 123, 53–61.

Kwan B.K., Cheung J.H., Law A.C., Cheung S.G. & Shin P.K. (2017) Conservation education program for threatened Asian horseshoe crabs: a step towards reducing community apathy to environmental conservation. *Journal for Nature Conservation*, 35, 53–65.*

Kyle C., Cavanaugh J.R., Kellner A.J., Forde D.S., Gruner J.D., Parker W., Rodriguez I. & Feller C. (2014) Poleward expansion of mangroves is a threshold response to decreased frequency of extreme cold events. *Proceedings of the National Academy of Sciences*, 111, 723–727.

Lambert G.I., Jennings S., Kaiser M.J., Davies T.W. & Hiddink J.G. (2014) Quantifying recovery rates and resilience of seabed habitats impacted by bottom fishing. *Journal of Applied Ecology*, 51, 1326–1336.

Langhamer O. & Wilhelmsson D. (2009) Colonisation of fish and crabs of wave energy foundations and the effects of manufactured holes–a field experiment. *Marine Environmental Research*, 68, 151–157.

Langhamer O. (2012) Artificial reef effect in relation to offshore renewable energy conversion: State of the art. *The Scientific World Journal*.

Lefebvre-Chalain H. (2007) Fifteen years of particularly sensitive sea areas: a concept in development. *Ocean & Coastal Law Journal*, 13, 47.

Leisher C., Mangubhai S., Hess S., Widodo H., Soekirman T., Tjoe S., Wawiyai S., Larsen S.N., Rumetna L., Halim A. & Sanjayan M. (2012) Measuring the benefits and costs of community education and outreach in marine protected areas. *Marine Policy*, 36, 1005–1011.

Leitão F.M.S. & Gaspar M.B. (2007) Immediate effect of intertidal non-mechanised cockle harvesting on macrobenthic communities: a comparative study. Scientia Marina, 71, 723–733.

Lenzi M., Palmieri R. & Porrello S. (2003) Restoration of the eutrophic Orbetello lagoon (Tyrrhenian Sea, Italy): water quality management. *Marine Pollution Bulletin,* 46, 1540–1548.

Lester S.E., Halpern B.S., Grorud-Colvert K., Lubchenco J., Ruttenberg B.I., Gaines S.D., Airamé S. & Warner R.R. (2009) Biological effects within no-take marine reserves: a global synthesis. *Marine Ecology Progress Series*, 384, 33–46.*

Levitt G.J., Anderson R.J., Boothroyd C.J.T. & Kemp F.A. (2002) The effects of kelp harvesting on its regrowth and the understorey benthic community at Danger Point, South Africa, and a new method of harvesting kelp fronds. *South African Journal of Marine Science,* 24, 71–85.

Lewis S.G. (2015) Bags and tags: randomized response technique indicates reductions in illegal recreational fishing of red abalone (*Haliotis rufescens*) in Northern California. *Biological Conservation,* 189, 72–77.*

Ley-Cooper K., De Lestang S., Phillips B.F. & Lozano-Álvarez E. (2014) An unfished area enhances a spiny lobster, *Panulirus argus*, fishery: implications for management and conservation within a Biosphere Reserve in the Mexican Caribbean. *Fisheries Management and Ecology*, 21, 264–274.*

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.

Liley D., Morris R.K.A., Cruickshanks K., Macleod C., Underhill-Day J., Brereton T. & Mitchell J., (2012) Identifying best practice in management of activities on Marine Protected Areas. Footprint Ecology/Bright Angel Consultants/MARINElife. Natural England Commissioned Reports, Number 108.

Lin D.T. & Bailey-Brock J.H. (2008) Partial recovery of infaunal communities during a fallow period at an open-ocean aquaculture. *Marine Ecology Progress Series,* 371, 65–72.*

Lin H.J., Shao K.T., Hsieh H.L., Lo W.T. & Dai X.X. (2009) The effects of system-scale removal of oyster-culture racks from Tapong Bay, southwestern Taiwan: model exploration and comparison with field observations. *ICES Journal of Marine Science*, 66, 797–810.*

Ling S.D. & Johnson C.R. (2012) Marine reserves reduce risk of climate-driven phase shift by reinstating size-and habitat-specific trophic interactions. *Ecological Applications,* 22, 1232–1245.*

Lipcius R.N., Stockhausen W.T. & Eggleston D.B. (2001) Marine reserves for Caribbean spiny lobster: empirical evaluation and theoretical metapopulation recruitment dynamics. *Marine and Freshwater Research,* 52, 1589–1598.*

Liu X.-S., Xu W.-Z., Cheung S.G. & Shin P.K.S. (2011) Response of meiofaunal community with special reference to nematodes upon deployment of artificial reefs and cessation of bottom trawling in subtropical waters, Hong Kong. *Marine Pollution Bulletin,* 63, 376–384.*

Lloret J., Zaragoza N., Caballero D. & Riera V. (2008) Impacts of recreational boating on the marine environment of Cap de Creus (Mediterranean Sea). *Ocean & Coastal Management*, 51, 749–754.

Loan M.E., Herron M., Akkurt R., Pomerantz A.E. & Schlumberger Technology Corp. (2018) Oil-based mud drill cutting cleaning for infrared spectroscopy. U.S. Patent Application 15/410,045.

Lozano-Álvarez E., Briones-Fourzán P., Álvarez-Filip L., Weiss H.M., Negrete-Soto F. & Barradas-Ortiz C. (2010) Influence of shelter availability on interactions between Caribbean spiny lobsters and moray eels: Implications for artificial lobster enhancement. *Marine Ecology Progress Series,* 400, 175–185.*

Lozano-Álvarez E., Meiners C. & Briones-Fourzán P. (2009) Ontogenetic habitat shifts affect performance of artificial shelters for Caribbean spiny lobsters. *Marine Ecology Progress Series*, 396, 85–97.*

Lück M. (2003) Education on marine mammal tours as agent for conservation—but do tourists want to be educated? *Ocean & Coastal Management*, 46, 943–956.

Luna B., Pérez C.V. & Sánchez-Lizaso J.L. (2009) Benthic impacts of recreational divers in a Mediterranean Marine Protected Area. *ICES Journal of Marine Science*, 66, 517–523.

Macleod C.K., Moltschaniwskyj N.A. & Crawford C.M. (2006) Evaluation of short-term fallowing as a strategy for the management of recurring organic enrichment under salmon cages. *Marine Pollution Bulletin*, 52, 1458–1466.*

Macleod C.K., Moltschaniwskyj N.A., Crawford C.M. & Forbes S.E. (2007) Biological recovery from organic enrichment: some systems cope better than others. *Marine Ecology Progress Series*, 342, 41–53.*

Magin C.M., Cooper S.P. & Brennan A.B. (2010). Non-toxic antifouling strategies. *Materials Today*, 13, 36-44.

Maguire J.A*.,* Knights A.M., O'Toole M., Burnell G., Crowe T.P., Ferns M., McDonough N., McQuaid N., O'Connor B., Doyle R. & Newell C. (2008) Management recommendations for sustainable exploitation of mussel seed in the Irish Sea. *Marine Environment and Health Series*, 31.

Major R.N., Taylor D.I., Connor S., Connor G. & Jeffs A.G. (2017) Factors affecting bycatch in a developing New Zealand scampi potting fishery. *Fisheries Research*, 186, 55–64.*

Manchester S.J. & Bullock J.M. (2000) The impacts of non-native species on UK biodiversity and the effectiveness of control. *Journal of Applied Ecology*, 37, 845–864.

Manríquez P.H. & Castilla J.C. (2001) Significance of marine protected areas in central Chile as seeding grounds for the gastropod *Concholepas concholepas*. *Marine Ecology Progress Series*, 215, 201–211.*

Marinho C.H., Giarratano E., Esteves J.L., Narvarte M.A. & Gil M.N. (2017) Hazardous metal pollution in a protected coastal area from Northern Patagonia (Argentina). *Environmental Science and Pollution Research*, 24, 6724–6735.

Marino II M.C., Juanes F. & Stokesbury K.D.E. (2007). Effect of closed areas on populations of sea star *Asterias* spp. on Georges Bank. *Marine Ecology Progress Series,* 347, 39–49.*

Maselko J., Bishop G. & Murphy P. (2013) Ghost fishing in the Southeast Alaska commercial Dungeness crab fishery. *North American Journal of Fisheries Management*, 33, 422–431.

Matsuoka T., Nakashima T. & Nagasawa N. (2005) A review of ghost fishing: scientific approaches to evaluation and solutions. *Fisheries Science*, 71, 691.

Mayor D. J., Solan M., Martinez I., Murray L., McMillan H., Paton G. I. & Killham K. (2008). Acute toxicity of some treatments commonly used by the salmonid aquaculture industry to Corophium volutator and Hediste diversicolor: Whole sediment bioassay tests. *Aquaculture*, 285, 102–108.

Mazzola A., Mirto S., La Rosa T., Fabiano M. & Danovaro R. (2000) Fish-farming effects on benthic community structure in coastal sediments: analysis of meiofaunal recovery. *ICES Journal of Marine Science*, 57, 1454–1461.*

McCay D.P.F., Peterson C.H., DeAlteris J.T. & Catena J. (2003) Restoration that targets function as opposed to structure: replacing lost bivalve production and filtration. *Marine Ecology Progress Series*, 264, 197–212.

McConnaughey R.A., Mier K.L. & Dew C.B. (2000) An examination of chronic trawling effects on soft-bottom benthos of the eastern Bering Sea. *ICES Journal of Marine Science,* 57, 1377–1388.*

McGann M., Alexander C.R. & Bay S.M. (2003) Response of benthic foraminifers to sewage discharge and remediation in Santa Monica Bay, California. *Marine Environmental Research*, 56, 299–342.

Mcleod E., Chmura G.L., Bouillon S., Salm R., Björk M., Duarte C.M., Lovelock C.E., Schlesinger W.H. & Silliman B.R. (2011) A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO_2. *Frontiers in Ecology and the Environment,* 9, 552–560.

Melton H.R., Smith J.P., Mairs H.L., Bernier R.F., Garland E., Glickman A.H., Jones F.V., Ray J.P., Thomas D. & Campbell J.A. (2004) *Environmental aspects of the use and disposal of non aqueous drilling fluids associated with offshore oil & gas operations.* In: SPE International Conference on Health, Safety, and Environment in Oil and Gas Exploration and Production. Society of Petroleum Engineers.

Melton H.R., Smith J.P., Martin C.R., Nedwed T.J., Mairs H.L. & Raught D.L. (2000) *Offshore discharge of drilling fluids and cuttings–a scientific perspective on public policy*. In Rio Oil and Gas Conference. Rio de Janeiro, Brazil.

Meyer D.L. & Townsend E.C. (2000) Faunal utilization of created intertidal eastern oyster (*Crassostrea virginica*) reefs in the southeastern United States. *Estuaries*, 23, 34–45

Milazzo M., Badalamenti F., Ceccherelli G. & Chemello R. (2004) Boat anchoring on Posidonia oceanica beds in a marine protected area (Italy, western Mediterranean): effect of anchor types in different anchoring stages. Journal of Experimental Marine Biology and Ecology, 299, 51–62.

Milazzo M., Chemello R., Badalamenti F., Camarda R. & Riggio S. (2002) The Impact of Human Recreational Activities in Marine Protected Areas: What Lessons Should Be Learnt in the Mediterranean Sea? *Marine Ecology,* 23, 280–290.

Miller K.A., Thompson K.F., Johnston P. & Santillo D. (2018) An overview of seabed mining including the current state of development, environmental impacts, and knowledge gaps. *Frontiers in Marine Science*, 4, 418.

Miller, R.J. (1976) North American crab fisheries: regulations and their rationales. *Fishery Bulletin*, 74, 623–633.

Mineur F., Cook E.J., Minchin D., Bohn K., MacLeod A. & Maggs C.A. (2012) Changing coasts: Marine aliens and artificial structures. Pages 198–243 in: *Oceanography and Marine Biology*. CRC Press.

Mirto S., Bianchelli S., Gambi C., Krzelj M., Pusceddu A., Scopa M., Holmer M. & Danovaro R. (2010) Fish-farm impact on metazoan meiofauna in the Mediterranean Sea: analysis of regional vs. habitat effects. *Marine Environmental Research*, 69, 38–47.

Moffa P.E. (Ed.). (1997) *The control and treatment of combined sewer overflows*. John Wiley & Sons.

Mohapatra A., Mohanty R.K., Mohanty S.K., Bhatta K.S. & Das N.R. (2007) Fisheries enhancement and biodiversity assessment of fish, prawn and mud crab in Chilika lagoon through hydrological intervention. *Wetlands Ecology and Management*, 15, 229–251.*

Moland E., Olsen E.M., Knutsen H., Garrigou P., Espeland S.H., Kleiven A.R., André C., & Knutsen J.A. (2013) Lobster and cod benefit from small-scale northern marine protected areas: inference from an empirical before–after control-impact study. *Proceedings of the Royal Society B: Biological Sciences,* 280, 1754.*

Molnar J.L., Gamboa R.L., Revenga C. & Spalding M.D. (2008) Assessing the global threat of invasive species to marine biodiversity. *Frontiers in Ecology and the Environment*, 6, 485–492.

Moran M.J. & Stephenson P.C. (2000) Effects of otter trawling on macrobenthos and management of demersal scalefish fisheries on the continental shelf of north-western Australia. *ICES Journal of Marine Science*, 57, 510–516.*

Morelli T.L., Daly C., Dobrowski S.Z., Dulen D.M., Ebersole J.L., Jackson S.T., Lundquist J.D., Millar C.I., Maher S.P., Monahan W.B. & Nydick K.R. (2016) Managing climate change refugia for climate adaptation. *PLoS One*, 11, p.e0159909.Gilbert A., Andréfouët S., Yan L. & Remoissenet G. (2006). The giant clam *Tridacna maxima* communities of three French Polynesia islands: comparison of their population sizes and structures at early stages of their exploitation. *ICES Journal of Marine Science*, 63, 1573–1589.

Morris A.S., Wilson S.M., Dever E.F. & Chambers R.M. (2011) A test of bycatch reduction devices on commercial crab pots in a tidal marsh creek in Virginia. *Estuaries and Coasts*, 34, 386–390.

Morrow D.R., & Larkin P.D. (2007) The challenges of pipeline burial. In *The Seventeenth International Offshore and Polar Engineering Conference*. International Society of Offshore and Polar Engineers.

Moschino V., Deppieri M. & Marin M.G. (2003) Evaluation of shell damage to the clam *Chamelea gallina* captured by hydraulic dredging in the Northern Adriatic Sea. *ICES Journal of Marine Science*, 60, 393–401.

Narvarte M. González R. Medina A. & Avaca M.S. (2011) Artisanal dredges as efficient and rationale harvesting gears in a Patagonian mussel fishery. Fisheries Research, 111, 108–115.*

Narvarte M., González R., Medina A., Avaca M.S., Ginsberg S. & Aliotta S. (2012) Short term impact of artisanal dredges in a Patagonian mussel fishery: Comparisons with commercial diving and control sites. *Marine Environmental Research,* 73, 53–61.*

Navarro-Barranco C. & Hughes L.E. (2015) Effects of light pollution on the emergent fauna of shallow marine ecosystems: Amphipods as a case study. *Marine Pollution Bulletin*, 94, 235–240.

Naylor E. (1965) Effects of heated effluents upon marine and estuarine organisms. Pages 63–103 in: *Advances in Marine Biology*). Academic Press.

Naylor R.L., Goldburg R.J., Primavera J.H., Kautsky N., Beveridge M.C., Clay J., Folke C., Lubchenco J., Mooney H. & Troell M. (2000) Effect of aquaculture on world fish supplies. *Nature*, 405, 1017.

Nelson KA., Leonard L.A., Posey M.H., Alphin T.D. & Mallin M.A. (2004) Using transplanted oyster (*Crassostrea virginica*) beds to improve water quality in small tidal creeks: a pilot study. *Journal of Experimental Marine Biology and Ecology*, 298, 347–368.

Newell R.C., Seiderer L.J., Simpson N.M., & Robinson J.E. (2004) Impacts of marine aggregate dredging on benthic macrofauna off the south coast of the United Kingdom. *Journal of Coastal Research*, 20, 115–125.

Nizinski M.S. (2007) Predation in subtropical soft-bottom systems: Spiny lobster and molluscs in Florida Bay. *Marine Ecology Progress Series,* 345, 185–197.*

Nobre A.M., Robertson-Andersson D., Neori A. & Sankar K. (2010) Ecological–economic assessment of aquaculture options: comparison between abalone monoculture and integrated multi-trophic aquaculture of abalone and seaweeds. *Aquaculture*, 306, 116–126.

Nuñez M.A., Kuebbing S., Dimarco R.D. & Simberloff D. (2012) Invasive species: to eat or not to eat, that is the question. *Conservation Letters*, 5, 334–341.

Öndes F., Kaiser M.J. & Murray L.G. (2017) Fish and invertebrate by-catch in the crab pot fishery in the Isle of Man, Irish Sea. *Journal of the Marine Biological Association of the United Kingdom*, 98, 1–13.

Ospar Commission. (2000) OSPAR Decision 2000/3 on the use of rganic-Phase Drilling Fluids (OPF) and the discharge of OPF-contaminated cuttings. OSPAR, Copenhagen, Denmark.

OSPAR Decision 98/3 (1998) On the Disposal of Disused Offshore Installations. In: Ministerial Meeting of the OSPAR Commission, OSPAR Convention for the protection of the marine environment of the North-East Atlantic, London, UK.

Pande A., MacDiarmid A.B., Smith P.J., Davidson R.J., Cole R.G., Freeman D., Kelly S. & Gardner J.P. (2008) Marine reserves increase the abundance and size of blue cod and rock lobster. *Marine Ecology Progress Series*, 366, 147–158.*

Parker L.M., Ross P.M. & O'Connor W.A. (2011) Populations of the Sydney rock oyster, *Saccostrea glomerata*, vary in response to ocean acidification. *Marine Biology*, 158,3, 689–697.

Parnell P.E., Lennert-Cody C.E., Geelen L., Stanley L.D. & Dayton P.K. (2005) Effectiveness of a small marine reserve in southern California. *Marine Ecology Progress Series*, 296, 39–52.*

Pasko S., Goldberg J., MacNeil C. & Campbell M. (2014) Review of harvest incentives to control invasive species. *Management of Biological Invasions*, 5, 263–277.

Patel A., Stamatakis S., Young S. & Friedheim J. (2007) *Advances in inhibitive water-based drilling fluids—can they replace oil-based muds?* In: International Symposium on Oilfield Chemistry. Society of Petroleum Engineers.

Payne C. & Sand P. (Eds.) (2011) *Gulf War Reparations and the UN Compensation Commission Environmental Liability*. Oxford University Press, Oxford.

Peterson C.H., Summerson H.C. & Luettich Jr.R.A. (1996) Response of bay scallops to spawner transplants: a test of recruitment limitation. *Marine Ecology Progress Series*, 132, 93–107.*

Pham C.K., Ramirez-Llodra E., Alt C.H., Amaro T., Bergmann M., Canals M., Davies J., Duineveld G., Galgani F., Howell K.L. & Huvenne V.A. (2014) Marine litter distribution and density in European seas, from the shelves to deep basins. *PloS One*, 9, p.e95839.

Phillips D.I. (1999). A new litter trap for urban drainage systems. *Water Science and Technology*, 39, 85–92.

Piazzi L., La Manna G., Cecchi E., Serena F. & Ceccherelli G. (2016) Protection changes the relevancy of scales of variability in coralligenous assemblages. *Estuarine, Coastal and Shelf Science,* 175, 62–69.*

Picken G., Curtis T. & Elliott A. (1997) An estimate of the cumulative environmental effects of the disposal in the deep sea of bulky wastes from the offshore oil and gas industry. In Offshore Europe. Society of Petroleum Engineers.

Piferer F., Beaumont A., Falguiere J-C., Flajshans M., Haffray P. & Colombo L. (2009) Polyploidy fish and shellfish: Production, biology, and applications to aquaculture for performance improvement and genetic containment. *Aquaculture*, 293, 125–156.

Pillans S., Pillans R.D., Johnstone R.W., Kraft P.G., Haywood M.D.E. & Possingham H.P. (2005) Effects of marine reserve protection on the mud crab *Scylla serrata* in a sex-biased fishery in subtropical Australia. *Marine Ecology Progress Series,* 295, 201–213.*

Pine M.K., Andrew J.G. & Radford C.A. (2012) Turbine sound may influence the metamorphosis behaviour of estuarine crab megalopae. *PLoS One*, 7, p. e51790

Pirker J.G. (2002) Demography, biomass production and effects of harvesting giant kelp *Macrocystis pyrifera* (Linnaeus) in Southern New Zealand.

Pitt N.R., Poloczanska E.S. & Hobday A.J. (2010) Climate-driven range changes in Tasmanian intertidal fauna. *Marine and Freshwater Research*, 61, 963–970.

Ponti M., Abbiati M. & Ceccherelli V.U. (2002) Drilling platforms as artificial reefs: distribution of macrobenthic assemblages of the "Paguro" wreck (northern Adriatic Sea). *ICES Journal of Marine Science*, 59, 316–323.*

Ponti M., Fava F., Perlini R.A., Giovanardi O. & Abbiati M. (2015) Benthic assemblages on artificial reefs in the northwestern Adriatic Sea: Does structure type and age matter*? Marine Environmental Research*, 104, 10–19.*

Popper A.N. & Hawkins A. (Eds.) (2012) *The Effects of Noise on Aquatic Life*, Springer Science + Business Media, LLC, New York

Price A.R.G., Donlan M.C., Sheppard C.R.C. & Munawar M. (2012) Environmental rejuvenation of the Gulf by compensation and restoration. *Aquatic Ecosystem Health & Management,* 15, 7–13.

Prior S. (2011) *Investigating the use of voluntary marine management in the protection of UK marine biodiversity.* Report to Wales Environment Link.

Qiao B., Chu J.C., Zhao P., Yu Y. & Li Y. (2002) Marine oil spill contingency planning. Journal of Environmental Sciences, 14, 102–107.

Raemaekers S., Hauck M., Bürgener M., Mackenzie A., Maharaj G., Plagányi É.E. & Britz P.J. (2011) Review of the causes of the rise of the illegal South African abalone fishery and consequent closure of the rights-based fishery. *Ocean & Coastal Management*, 54, 433–445.

Rainbow P.S. (2017) Heavy metal levels in marine invertebrates. In *Heavy metals in the marine environment*, 67–79. CRC press.

Rawlins B.G., Ferguson A.J., Chilton P.J., Arthurton R.S., Rees J.G. & Baldock J.W. (1998) Review of agricultural pollution in the Caribbean with particular emphasis on small island developing states. *Marine Pollution Bulletin*, 36, 658–668.

Rees H.L., Waldock R., Matthiessen P. & Pendle M.A. (1999) Surveys of the epibenthos of the Crouch Estuary (UK) in relation to TBT contamination. *Journal of the Marine Biological Association of the United Kingdom*, 79, 209–223.*

Reine K., Clarke D., Ray G. & Dickerson C. (2013) Fishery resource utilization of a restored estuarine borrow pit: A beneficial use of dredged material case study. *Marine Pollution Bulletin*, 73, 115–128.*

Reise K., Gollasch S. & Wolff W.J. (1998) Introduced marine species of the North Sea coasts. *Helgoländer Meeresuntersuchungen*, 52, 219

Revill A.S. & Jennings S. (2005) The capacity of benthos release panels to reduce the impacts of beam trawls on benthic communities. *Fisheries Research*, 75, 73–85.*

Roach A.C. & Wilson S.P. (2009). Ecological impacts of tributyltin on estuarine communities in the Hastings River, NSW Australia. *Marine Pollution Bulletin*, 58, 1780–1786.

Roach M., Cohen M., Forster R., Revill A.S., Johnson M. & Handling editor: Steven Degraer. (2018) The effects of temporary exclusion of activity due to wind farm construction on a lobster (*Homarus gammarus*) fishery suggests a potential management approach. *ICES Journal of Marine Science*, 75, 1416–1426.*

Rogers-Bennett L. & Pearse J.S. (2001) Indirect benefits of marine protected areas for juvenile abalone. *Conservation Biology*, 15, 642–647.*

Rogers-Bennett L., Haaker P.L., Karpov K.A. & Kushner D.J. (2002) Using spatially explicit data to evaluate marine protected areas for abalone in southern California. *Conservation Biology*, 16, 1308–1317.*

Rogers-Bennett L., Hubbard K.E. & Juhasz C.I. (2013) Dramatic declines in red abalone populations after opening a “de facto” marine reserve to fishing: Testing temporal reserves. *Biological Conservation,* 157, 423–431.*

Romano C., Fanelli E., D'Anna G., Pipitone C., Vizzini S., Mazzola A. & Badalamenti F. (2016) Spatial variability of soft-bottom macrobenthic communities in northern Sicily (Western Mediterranean): Contrasting trawled vs. untrawled areas. *Marine Environmental Research,* 122, 113–125.*

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.

Ruiz G.M., Carlton J.T., Grosholz E.D. & Hines A.H. (1997) Global invasions of marine and estuarine habitats by non-indigenous species: mechanisms, extent, and consequences. *American Zoologist*, 37, 621–632.

Russell G., Hawkins S.J., Evans L.C., Jones H.D. & Holmes G.D. (1983) Restoration of a disused dock basin as a habitat for marine benthos and fish. *Journal of Applied Ecology*, 43–58.

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.

Santos J., Herrmann B., Mieske B., Krag L. A., Haase S. & Stepputtis D. (2018) The efficiency of sieve-panels for bycatch separation in *Nephrops* trawls. *Fisheries Management and Ecology*, 25, 464–473.

Saphier A.D. & Hoffmann T.C. (2005) Forecasting models to quantify three anthropogenic stresses on coral reefs from marine recreation: Anchor damage, diver contact and copper emission from antifouling paint. *Marine Pollution Bulletin*, 51, 590–598.

Sarà G., Scilipoti D., Milazzo M. & Modica A. (2006) Use of stable isotopes to investigate dispersal of waste from fish farms as a function of hydrodynamics. *Marine Ecology Progress Series*, 313, 261–270.

Sardá R., Pinedo S., Gremare A. & Taboada S. (2000) Changes in the dynamics of shallow sandy-bottom assemblages due to sand extraction in the Catalan Western Mediterranean Sea. *ICES Journal of Marine Science,* 57, 1446–1453.

Schejter L., Bremec C.S. & Hernández D. (2008) Comparison between disturbed and undisturbed areas of the Patagonian scallop (*Zygochlamys patagonica*) fishing ground “Reclutas” in the Argentine Sea. *Journal of Sea Research*, 60, 193–200.*

Schejter L., Bremec C.S. & Hernández D. (2009) Erratum to “Comparison between disturbed and undisturbed areas of the Patagonian scallop (*Zygochlamys patagonica*) fishing ground “Reclutas” in the Argentine Sea” [J. Sea Research 60/3 (2008) 193]. *Journal of Sea Research* 61, 275.

Schmalenbach I., Mehrtens F., Janke M. & Buchholz F. (2011) A mark-recapture study of hatchery-reared juvenile European lobsters, *Homarus gammarus*, released at the rocky island of Helgoland (German Bight, North Sea) from 2000 to 2009. *Fisheries Research*, 108, 22–30.*

Schoeman D.S., Cockcroft A.C., Van Zyl D.L. & Goosen P.C. (2002) Changes to regulations and the gear used in the South African commercial fishery for *Jasus lalandii. South African Journal of Marine Science*, 24, 365–369.

Schroeter S.C., Reed D.C. & Raimondi P.T. (2015) Effects of reef physical structure on development of benthic reef community: A large-scale artificial reef experiment. *Marine Ecology Progress Series,* 540, 43–55.*

Schwinghamer P., Gordon Jr D.C., Rowell T.W., Prena J., McKeown D.L., Sonnichsen G. & Guigné J.Y. (1998) Effects of experimental otter trawling on surficial sediment properties of a sandy-bottom ecosystem on the Grand Banks of Newfoundland. *Conservation Biology*, 12, 1215–1222.

Sciberras M., Hiddink J.G., Jennings S., Szostek C.L., Hughes K.M., Kneafsey B., Clarke L.J., Ellis N., Rijnsdorp A.D., McConnaughey R.A., Hilborn R., Collie J.S., Pitcher C.R., Amoroso R.O., Parma A.M., Suuronen P. & Kaiser M.J. (2018) Response of benthic fauna to experimental bottom fishing: a global meta-analysis. *Fish & Fisheries*, 19, 698–715.

Sciberras M., Hinz H., Bennell J.D., Jenkins S.R., Hawkins S.J. & Kaiser M.J. (2013) Benthic community response to a scallop dredging closure within a dynamic seabed habitat. *Marine Ecology Progress Series,* 480, 83–98.*

Sciberras M., Jenkins S.R., Kaiser M.J., Hawkins S.J. & Pullin A.S. (2013) Evaluating the biological effectiveness of fully and partially protected marine areas. *Environmental Evidence*, 2, 4.

Scott K., Harsanyi P. & Lyndon A.R. (2019) Understanding the effects of electromagnetic field emissions from Marine Renewable Energy Devices (MREDs) on the commercially important edible crab, *Cancer pagurus* (L.). *Frontier in Marine Science Conference Abstract: IMMR'18 | International Meeting on Marine Research 2018.*

Scyphers S.B., Powers S.P. & Heck K.L. (2015) Ecological value of submerged breakwaters for habitat enhancement on a residential scale. *Environmental Management*, 55, 383–391.*

Serrano A. Rodríguez-Cabello C. Sánchez F. Velasco F. Olaso I. & Punzón A. (2011) Effects of anti-trawling artificial reefs on ecological indicators of inner shelf fish and invertebrate communities in the Cantabrian Sea (southern Bay of Biscay). *Journal of the Marine Biological Association of the United Kingdom,* 91, 623–633.*

Shadizadeh S.R., Majidaie S. & Zoveidavianpoor M. (2011) Investigation of drill cuttings reinjection: Environmental management in Iranian Ahwaz Oilfield. *Petroleum Science and Technology*, 29, 1093–1103

Shears N.T. & Babcock R.C. (2003) Continuing trophic cascade effects after 25 years of no-take marine reserve protection. *Marine Ecology Progress Series,* 246, 1–16.*

Shears N.T., Grace R.V., Usmar N.R., Kerr V. & Babcock R.C. (2006) Long-term trends in lobster populations in a partially protected vs. no-take Marine Park. *Biological Conservation*, 132, 222–231.*

Sheehan E.V., Bridger D., Cousens S.L. & Attrill M.J. (2015) Testing the resilience of dead maerl infaunal assemblages to the experimental removal and re-lay of habitat. *Marine Ecology Progress Series*, *535*, 117–128.*

Sheehan E.V., Cousens S.L., Nancollas S.J., Stauss C., Royle J. & Attrill M.J. (2013) Drawing lines at the sand: Evidence for functional vs. visual reef boundaries in temperate Marine Protected Areas. *Marine Pollution Bulletin*, 76, 194–202.*

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.

Simberloff D., Alexander J., Allendorf F., Aronson J., Antunes P.M., Bacher S., Bardgett R., Bertolino S., Bishop M., Blackburn T.M. & Blakeslee A. (2011) Non-natives: 141 scientists object. *Nature*, 475, 7354.

Simonini R., Ansaloni I., Bonini P., Grandi V., Graziosi F., Iotti M., Massamba-N'Siala G., Mauri M., Montanari G., Preti M. & De Nigris N. (2007) Recolonization and recovery dynamics of the macrozoobenthos after sand extraction in relict sand bottoms of the Northern Adriatic Sea. *Marine Environmental Research*, 64, 574–589.*

Smale D.A. & Wernberg T. (2013. Extreme climatic event drives range contraction of a habitat-forming species. Proceedings of the Royal Society B: Biological Sciences, 280, 1754.

Smiley P.C. & Allred B.J. (2011) Differences in aquatic communities between wetlands created by an agricultural water recycling system. *Wetlands Ecology and Management*, 19, 495–505.

Smith B.E. Collie J.S. & Lengyel N.L. (2013) Effects of chronic bottom fishing on the benthic epifauna and diets of demersal fishes on northern Georges Bank. *Marine Ecology Progress Series*, 472, 199–217.*

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.

Soetaert M., Lenoir H. & Verschueren B. (2016) Reducing bycatch in beam trawls and electrotrawls with (electrified) benthos release panels. *ICES Journal of Marine Science*, 73, 2370–2379.*

Sonnenholzner J.I., Ladah L.B. & Lafferty K.D. (2009) Cascading effects of fishing on Galapagos rocky reef communities: Reanalysis using corrected data. *Marine Ecology Progress Series,* 375, 209–218.*

Spagnolo A., Cuicchi C., Punzo E., Santelli A., Scarcella G. & Fabi G. (2014) Patterns of colonization and succession of benthic assemblages in two artificial substrates. *Journal of Sea Research*, 88, 78–86.*

Stagnol D., Michel R. & Davoult D. (2016) Unravelling the impact of harvesting pressure on canopy-forming macroalgae. *Marine and Freshwater Research*, 67, 153–161.

Stark J.S., Johnstone G.J. & Riddle M.J. (2014) A sediment mesocosm experiment to determine if the remediation of a shoreline waste disposal site in Antarctica caused further environmental impacts. Marine Pollution Bulletin, 89, 284–295.*

Stark J.S., Snape I. & Riddle M.J. (2006) Abandoned Antarctic waste disposal sites: monitoring remediation outcomes and limitations at Casey Station. Ecological Management & Restoration, 7, 21–31.

Steimle F., Foster K., Kropp R. & Conlin B. (2002) Benthic macrofauna productivity enhancement by an artificial reef in Delaware Bay, USA. *ICES Journal of Marine Science*, 59, 100–105.*

Steppe C.N., Fredriksson D.W., Wallendorf L., Nikolov M. & Mayer R. (2016) Direct setting of *Crassostrea virginica* larvae in a tidal tributary: applications for shellfish restoration and aquaculture. *Marine Ecology Progress Series*, 546, 97–112.*

Stevens B.G. (1996) Crab bycatch in pot fisheries. *Solving bycatch: considerations for today and tomorrow*, 151–158.

Stierhoff K.L., Neuman M. & Butler J.L. (2012) On the road to extinction? Population declines of the endangered white abalone, *Haliotis sorenseni*. *Biological Conservation*, 152, 46–52.

Stoner A.W. & Ray M. (1996) Queen conch, *Strombus gigas*, in fished and unfished locations of the Bahamas: Effects of a marine fishery reserve on adults, juveniles, and larval production. *Fishery Bulletin*, 94, 551–565.*

Sumaila U.R. & Pauly D. (2007) All fishing nations must unite to cut subsidies. *Nature*, 450, 945.

Sun P., Liu X., Tang Y., Cheng W., Sun R., Wang X., Wan R., Heino M. & Handling editor: Jonathan Grabowski (2017) The bio-economic effects of artificial reefs: mixed evidence from Shandong, China. *ICES Journal of Marine Science*, 74, 2239–2248.*

Swan K.D., McPherson J.M., Seddon P.J. & Moehrenschlager A. (2016) Managing marine biodiversity: the rising diversity and prevalence of marine conservation translocations. *Conservation Letters*, 9, 239–251.

Szymelfenig M., Kotwicki L. & Graca B. (2006) Benthic re-colonization in post-dredging pits in the Puck Bay (Southern Baltic Sea). *Estuarine, Coastal and Shelf Science*, 68, 489–498.

Takeuchi I., Takahashi S. & Tanabe S. (2004) Decline of butyltin levels in *Caprella* spp. (Crustacea: Amphipoda) inhabiting the *Sargassum* community in Otsuchi Bay, Japan from 1994 to 2001. *Journal of the Marine Biological Association of the United Kingdom*, 84, 911–918.*

Talman S.G., Norkko A., Thrush S.F. & Hewitt J.E. (2004) Habitat structure and the survival of juvenile scallops *Pecten novaezelandiae*: Comparing predation in habitats with varying complexity. *Marine Ecology Progress Series*, 269, 197–207.*

Teck S.J., Lorda J., Shears N.T., Bell T.W., Cornejo-Donoso J., Caselle J.E., Hamilton S.L. & Gaines S.D., (2017) Disentangling the effects of fishing and environmental forcing on demographic variation in an exploited species. *Biological Conservation*, 209, 488–498.*

Tettelbach S.T., Barnes D., Aldred J., Rivara G., Bonal D., Weinstock A., Fitzsimons-Diaz C., Thiel J., Cammarota M.C., Stark A. & Wejnert K. (2011) Utility of high-density plantings in bay scallop, *Argopecten irradians irradians,* restoration. *Aquaculture International*, 19, 715–739.*

Tettelbach S.T., Peterson B.J., Carroll J.M., Furman B.T., Hughes S.W.T., Havelin J., Europe J.R., Bonal D.M., Weinstock A.J. & Smith C.F. (2015) Aspiring to an altered stable state: Rebuilding of bay scallop populations and fisheries following intensive restoration. *Marine Ecology Progress Series*, 529, 121–136.*

Tettelbach S.T., Peterson B.J., Carroll J.M., Hughes S.W.T., Bonal D.M., Weinstock A.J., Europe J.R., Furman B.T. & Smith C.F. (2013) Priming the larval pump: Resurgence of bay scallop recruitment following initiation of intensive restoration efforts. *Marine Ecology Progress Series*, 478, 153–172.*

Tewfik A., Babcock E.A., Gibson J., Perez V.R.B. & Strindberg S. (2017) Benefits of a replenishment zone revealed through trends in focal species at Glover's Atoll, Belize. *Marine Ecology Progress Series,* 580, 37–56.*

Theile S. (2001) *Queen conch fisheries and their management in the Caribbean*. Brussels: TRAFFIC Europe.

Thelen B.A. & Thiet R.K. (2009) Molluscan community recovery following partial tidal restoration of a New England estuary. *Restoration Ecology,* 17, 695–703.*

Thiet R.K., Kidd E., Wennemer J.M. & Smith S.M. (2014) Molluscan community recovery in a New England back-barrier salt marsh lagoon 10 years after partial restoration. *Restoration Ecology*, 22, 447–455.*

Thresher R., Grewe P., Patil J.G., Whyard S., Templeton C.M., Chaimongol A., Hardy C.M., Hinds L.A. & Dunham R. (2009) Development of repressible sterility to prevent the establishment of feral populations of exotic and genetically modified animals. *Aquaculture*, 290, 104–109.

Thrush S.F., Hewitt J.E., Cummings V.J., Dayton P.K., Cryer M., Turner S.J., Funnell G.A., Budd R.G., Milburn C.J. & Wilkinson M.R. (1998) Disturbance of the marine benthic habitat by commercial fishing: impacts at the scale of the fishery. *Ecological Applications*, 8, 866–879.

Tracey S.R. & Lyle J.M. (2011) Linking scallop distribution and abundance with fisher behaviour: implication for management to avoid repeated stock collapse in a recreational fishery. *Fisheries Management and Ecology*, 18, 221–232.*

Trannum H.C., Setvik Å., Norling K. & Nilsson H.C. (2011) Rapid macrofaunal colonization of water-based drill cuttings on different sediments. *Marine Pollution Bulletin*, 62, 2145–2156.

Troell M., Joyce A., Chopin T., Neori A., Buschmann A.H. & Fang J.G. (2009). Ecological engineering in aquaculture—potential for integrated multi-trophic aquaculture (IMTA) in marine offshore systems. *Aquaculture*, 297, 1–9.

Tuya F.C., Soboil M.L. & Kido J. (2000) An assessment of the effectiveness of marine protected areas in the San Juan Islands, Washington, USA. *ICES Journal of Marine Science*, 57, 1218–1226.*

Twist B.A., Hepburn C.D. & Rayment W.J. (2016) Distribution of the New Zealand scallop (*Pecten novaezealandiae*) within and surrounding a customary fisheries area. *ICES Journal of Marine Science*, 73, 384–393.*

van der Wal D., Forster R.M., Rossi F., Hummel H., Ysebaert T., Roose F. & Herman P.M. (2011) Ecological evaluation of an experimental beneficial use scheme for dredged sediment disposal in shallow tidal waters. *Marine Pollution Bulletin,* 62, 99–108.*

Van Marlen B., Bergman M.J.N., Groenewold S. & Fonds M. (2005) New approaches to the reduction of non-target mortality in beam trawling. *Fisheries Research,* 72, 333–345.

Van Marlen B., Wiegerinck J.A.M., van Os-Koomen E. & van Barneveld E. (2014) Catch comparison of flatfish pulse trawls and a tickler chain beam trawl. *Fisheries Research*, 151, 57–69.*

van Oppen M.J., Oliver J.K., Putnam H.M. & Gates R.D. (2015) Building coral reef resilience through assisted evolution. *Proceedings of the National Academy of Sciences*, 112, 2307–2313.

Vanaverbeke J. & Vincx M. (2008) Short-term changes in nematode communities from an abandoned intense sand extraction site on the Kwintebank (Belgian Continental Shelf) two years post-cessation. *Marine Environmental Research*, 66, 240–248.*

Vare L.L., Baker M.C., Howe J.A., Levin L.A., Neira C., Ramirez-Llodra E.Z., Reichelt-Brushett A., Rowden A.A., Shimmield T.M., Simpson S.L. & Soto E.H. (2018). Scientific considerations for the assessment and management of mine tailings disposal in the deep sea. *Frontiers in Marine Science*, 5, 17.

Veale L.O. Hill A.S. Hawkins S.J. & Brand A.R. (2001) Distribution and damage to the by-catch assemblages of the northern irish sea scallop dredge fisheries. *Journal of the Marine Biological Association of the United Kingdom,* 81, 85–96.*

Versar (1999). *Biological sampling for dredged holes in Barnegat Bay*, Ocean County, NJ. Data Report prepared for the US Army Corps of Engineers, Philadelphia District, Philadelphia, PA.

Villamor A. & Becerro M.A. (2012) Species, trophic, and functional diversity in marine protected and non-protected areas. *Journal of Sea Research*, 73, 109–116.*

Visser, R. & van der Meer, J. (2008) Immediate displacement of the seabed during Subsea Rock Installation (SRI). *Terra et Aqua*, 110.

Vitaliano J.J., Fromm S.A., Packer D.B., Reid R.N. & Pikanowski R.A. (2007) Recovery of benthic macrofauna from sewage sludge disposal in the New York Bight. *Marine Ecology Progress Series*, 342, 27–40.*

Waddy S. L., Burridge L.E., Hamilton M.N., Mercer S.M., Aiken D.E. & Haya K. (2002) Emamectin benzoate induces molting in American lobster, *Homarus americanus*. *Canadian Journal of Fisheries and Aquatic Sciences*, *59*, 1096–1099.

Wade O. Revill A.S. Grant A. & Sharp M. (2009) Reducing the discards of finfish and benthic invertebrates of UK beam trawlers. *Fisheries Research,* 97, 140–147.*

Waldock M.J., Waite M.E. & Thain J.E. (1988) Inputs of tbt to the marine environment from shipping activity in the U.K., *Environmental Technology Letters,* 9, 999–1010.

Waldock R., Rees H.L., Matthiessen P. & Pendle M.A. (1999) Surveys of the benthic infauna of the Crouch Estuary (UK) in relation to TBT contamination. *Journal of the Marine Biological Association of the United Kingdom*, 79, 225–232.*

Walker B.K., Henderson B. & Spieler R.E. (2002) Fish assemblages associated with artificial reefs of concrete aggregates or quarry stone offshore Miami Beach, Florida, USA. *Aquatic Living Resources*, 15, 95–105.*

Watling L., & Norse E.A. (1998) Disturbance of the seabed by mobile fishing gear: a comparison to forest clearcutting. *Conservation Biology*, 12, 1180–1197.

Waye-Barker G.A., McIlwaine P., Lozach S. & Cooper K.M. (2015) The effects of marine sand and gravel extraction on the sediment composition and macrofaunal community of a commercial dredging site (15 years post-dredging). *Marine Pollution Bulletin*, 99, 207–215.*

Welles L.K. (2003) Comment: Due to loopholes in the Clean Water Act, what can a state do to combat cruise ship discharge of sewage and gray water. *Ocean & Coastal Law Journal*, 9, 99.

Werschkun B., Banerji S., Basurko O.C., David M., Fuhr F., Gollasch S., Grummt T., Haarich M., Jha A.N., Kacan S. & Kehrer A. (2014) Emerging risks from ballast water treatment: The run-up to the International Ballast Water Management Convention. *Chemosphere*, 112, 256–266.

West R.J. (2011) Impacts of recreational boating activities on the seagrass *Posidonia* in SE Australia. *Wetlands (Australia),* 26, 3–13.

Weston D.P. (2000). Ecological effects of the use of chemicals in aquaculture. Pages 23–30, in: J.R. Arthur, C.R. Lavilla-Pitogo, & R.P. Subasinghe (Eds.) *Use of Chemicals in Aquaculture in Asia : Proceedings of the Meeting on the Use of Chemicals in Aquaculture in Asia.* 20-22 May 1996. . Tigbauan, Iloilo, Philippines: Aquaculture Department, Southeast Asian Fisheries Development Center.

White H.K., Hsing P.Y., Cho W., Shank T.M., Cordes E.E., Quattrini A.M., Nelson R.K., Camilli R., Demopoulos A.W., German C.R. & Brooks J.M. (2012) Impact of the Deepwater Horizon oil spill on a deep-water coral community in the Gulf of Mexico. Proceedings of the National Academy of Sciences, 109, 20303–20308.

Whitfield P., Kenworthy W., Hammerstrom K. & Fonseca M. (2002) The Role of a Hurricane in the Expansion of Disturbances Initiated by Motor Vessels on Seagrass Banks. *Journal of Coastal Research*, 86–99.

Wilbur A.E., Seyoum S., Bert T.M. & Arnold W.S. (2005) A genetic assessment of bay scallop (*Argopecten irradians*) restoration efforts in Florida's Gulf of Mexico coastal waters (USA). *Conservation Genetics*, 6, 111–122.*

Wootton E.C., Woolmer A.P., Vogan C.L., Pope E.C., Hamilton K. M. & Rowley A.F. (2012) Increased disease calls for a cost-benefits review of marine reserves. *PLoS One*, 7, e51615.*

Wright J.T., Dworjanyn S.A., Rogers C.N., Steinberg P.D., Williamson J.E. & Poore A.G. (2005) Density-dependent sea urchin grazing: differential removal of species, changes in community composition and alternative community states. *Marine Ecology Progress Series*, 298, 143–156.

Wu R.S.S., Lam K.S., MacKay D.W., Lau T.C. & Yam V. (1994) Impact of marine fish farming on water quality and bottom sediment: A case study in the sub-tropical environment. *Marine Environmental Research,* 38, 115–145.

Yamamoto, T., Harada, K., Kim, K.H., Asaoka, S., & Yoshioka, I. (2013) Suppression of phosphate release from coastal sediments using granulated coal ash. *Estuarine, Coastal and Shelf Science*, 116, 41–49.*

Yamochi S. & Oda K. (2002) An attempt to restore suitable conditions for demersal fishes and crustaceans in the Port of Sakai-Semboku, north Osaka Bay, Japan. *Aquatic Ecology*, 36, 67–83.*

Yap H.T. (2009) Local changes in community diversity after coral transplantation. *Marine Ecology Progress Series*, 374, 33–41.*

Yip T.L., Talley W.K. & Jin D. (2011) The effectiveness of double hulls in reducing vessel-accident oil spillage. *Marine Pollution Bulletin,* 62, 2427–2432.

Zabin C.J., Wasson K. & Fork S. (2016) Restoration of native oysters in a highly invaded estuary. *Biological Conservation*, 202, 78–87.*

Zajicek P., Goodwin, A.E., & Weier, T. (2011) Triploid grass carp: Triploid induction, sterility, reversion, and certification. *North American Journal of Fisheries Management,* 31, 614–618.

Zeppel H. & Muloin S. (2008) Conservation benefits of interpretation on marine wildlife tours. *Human Dimensions of Wildlife*, 13, 280–294.

Zeppel H. (2008) Education and conservation benefits of marine wildlife tours: Developing free-choice learning experiences. *The Journal of Environmental Education*, 39, 3–18.

Zhulay I., Reiss K. & Reiss H. (2015) Effects of aquaculture fallowing on the recovery of macrofauna communities. *Marine Pollution Bulletin,* 97, 381–390.

Zupan M., Bulleri F., Evans J., Fraschetti S., Guidetti P., Garcia-Rubies A., Sostres M., Asnaghi V., Caro A., Deudero S. & Goñi R. (2018) How good is your marine protected area at curbing threats? *Biological Conservation*, 221, 237–245.

Appendix 1. Glossary of terms

Amphipod: Members of the phylum Crustacea. Mostly marine. They are small organisms resembling shrimps.

Biofouling: Also known as biological fouling. The unwanted accumulation of organisms on surfaces. In the marine environment, submerged or wet human-made structures (for instance aquaculture cages; vessel hulls; pontoons) can be colonised by organisms, which can lead to negative consequences.

Biogenic habitat: A habitat created by a living organism (a biogenic species) through its existence or behaviour. For example: coral reefs, oyster reefs, seagrass meadows, kelp forest, mussel beds. As compared to habitats formed through hydrological or geological processes (such as sandy beaches, bedrock, canyons, deep-sea vents, etc…).

Biogenic species: A species which has the ability, through its existence or behaviour, to create a new habitat/environment/ecosystem for other organisms. For example: certain coral species are biogenic and can create coral reefs; seagrass species forms meadows; certain oyster species can create ouster reefs.

Biological production: The total amount of biomass produced or regenerated by all living organisms in an area. This include primary production (by autotrophs such as algae and plants) and secondary production (by consumers).

Bycatch: This term is loosely used, and its exact meaning may vary with context. In this synopsis, we typically refrained from using this term.

Bycatch reduction device: Commonly referred to in the literature as BRDs. A general name used in fisheries to refer to a suite of net modifications and/or devices used on trawl nets to increase selectivity and reduce the amount of accidental unwanted catch by allowing them to 'escape' (for instance through holes, escape zones, or sections of the trawl net with bigger or different mesh geometry). BRDs tend to let species and organisms escape that are smaller than those targeted by the fishery.

Cephalopod: Members of the phylum Mollusca, they are exclusively marine and include squids, octopus, cuttlefish and nautilus. Several species are of commercial importance.

Codend: The narrow end part of a fishing trawl net where the catch is retained.

Cnidarian: Any members of the phylum Cnidaria. Mostly marine, they include: corals, sea anemones, jellyfish, hydroids, sea pens, sea whips, and sea fans.

Commercial catch: During fishing, this is the portion of the catch that is retained and has some economic value. It includes the species directly targeted by the fishery as well as other species of commercial value that are accidentally caught.

Crustacean: Any members of the subphylum Crustacean within the phylum Arthropoda. In majority marine, they include (but are not limited to): crabs, lobsters, prawns/shrimps, amphipods, barnacles, etc.

Discard: During fishing, this is the portion of the unwanted catch that is not retained and returned to the sea.

Echinoderm: Any members of the phylum Echinodermata. They include: starfish, brittle stars, sea urchins, sea cucumbers, and sea lilies (also known as crinoids).

Energy flow: A measure used to quantify the relative importance of different species of groups of species within the community trophic structure.

Epifauna: The animals that live on the surface of the sediments.

Gastropod: Also known as snails and slugs. Gastropods are members of the phylum Mollusca. In the marine environment, they include sea snails and nudribranchs.

Infauna: The animals that live inside the sediments.

Mollusc: Any members of the phylum Mollusca, the second-largest phylum of invertebrates. Mostly marine, they include (but are not limited to): cephalopods (such as squid, octopus and cuttlefish), gastropods (snails and nudibranchs), bivalves (such as oysters, mussels, and clams), chitons, tusk shells, etc.

Nematode: Also known as roundworms, nematodes are members of the phylum Nematoda. In the marine environment, they typically live in the sediments.

Polychaete: Also known as bristle worms, polychaetes are members of the class Polychaeta within the phylum Annelida. Mostly marine, they are often found in the sediments, but can also build their own tubes or live freely.

Sessile: Unable to move or with very limited mobility. Sessile animals and plants are typically attached to surfaces or live buried inside the sediment.

Shellfish: A commonly used term for commercially important species of aquatic organisms that have a shell or exoskeleton. They include (but are not limited to): molluscs such as oysters, mussels, abalone, winkles; crustaceans such as crabs, lobsters, prawns/shrimps; and echinoderms such as sea urchins.

Sievenet: A cone-shaped net (funnel-like device) inserted into standard fishing trawl nets, which directs unwanted catch to an escape hole in the body of the trawl. The idea is that the target species go over the hole in the net, while non-target can escape through the release hole. This 'bycatch reduction device' is based on the separator panel principle. It is not made of rigid material and therefore it is more acceptable to fishers than a rigid sorting grid.

Tunicates: Any members of the sub-phylum Tunicata (Phylum Vertebrata)**.** These include members of the sub-phylum Ascidiacea, commonly known as sea squirts.

Turtle excluder device: Commonly referred to in the literature as TEDs. A general name used in fisheries to refer to a suite of modifications and/or devices used on trawl nets to increase selectivity and reduce the amount of accidental unwanted catch by preventing organisms from entering the net and/or codend (for instance by fitting a sorting grid at the entrance of the codend). TEDs tend to prevent the entry of species and organisms larger than those targeted by the fishery. Originally developed to reduce the accidental catch of turtles, they are now widely used to prevent the catch of many large marine species.

Unwanted catch: During fishing, this is the portion of the catch caught in the net that is not directly targeted by the fishery. It includes unwanted non-commercial organisms ('discards'), and other 'bycatch' such as undersized individuals of the target species and commercial species which are not the main target of the fishery.

Zoanthid: Members of the order Zoantharia/Zoanthidae within the phylum Cnidaria. Resembles corals.

Appendix 2: Literature searched for the Subtidal Benthic Invertebrate Synopsis

A total of 270 journals were searched:

a) Journals directly relevant (25):

† signifies that the authors of this synopsis undertook parts or all of the systematic searches for the journal.
ᴵ signifies that a keyword search strategy was used instead to the subject-wide evidence synthesis method. See Appendix 3 for a full description of the search strings used.

JOURNAL	Volume/Year searched
African Journal of Marine Science†	Vol. 1 (1983) - Vol. 39 (2017)
Aquatic Conservation: Marine and Freshwater Ecosystems†	Vol. 1 (1991) - Vol. 27 (2017)
Aquatic Ecology (Springer)	Vol. 2 Issue 2 - Vol. 50 Issue 4 (2016)
Aquatic Ecosystem Health & Management	Vol. 1 Issue 1 - Vol. 19 Issue 4 (2016)
Aquatic Invasions	Vol. 1 (2006) - Vol. 11 (2016)
Aquatic Living Resources = Ressources Vivantes Aquatiques	Vol. 1 Issue 1 (1988) - Vol. 29 Issue 4 (2016)
Canadian Journal of Fisheries and Aquatic Sciences	Vol. 1 (1901) - Vol. 69 (2012)
Estuarine, Coastal and Shelf Science†	Keyword search 2000 - 2017ᴵ
Fish and Fisheries	Vol. 1 Issue 1 (2000) - Vol. 19 Issue 6 (2018)
Fisheries Management and Ecology	Vol. 1 Issue 1 (1994) - Vol. 25 Issue 6 (2018)
Fisheries Research†	Keyword search 2000 - 2017ᴵ
Hydrobiologia†	1995 - 2017
ICES Journal of Marine Science†	2000 - 2018
Journal of Sea Research (formerly Netherlands Journal of Sea Research)†	Vol. 1 (1961) - Vol. 129 (2017)
Journal of the Marine Biological Association of the United Kingdom†	Vol. 1 Issue 1 (1887) - Vol. 86 Issue 6 (2006) + keyword search 2000 - 2017
Journal of Wetlands Ecology	Vol. 1 (2008) - Vol. 6 (2012)
Journal of Wetlands Environmental Management	Vol. 1 (2012) - Vol. 4 (2016)
Limnologica - Ecology and Management of Inland Waters	Vol. 29 (1999) - Vol. 65 (2017)
Mangroves and Saltmarshes (Springer)	Vol. 1 (1996) – Vol. 3 (1999)
Marine Ecological Progress Series†	Keyword search 2010 - 2017ᴵ
Marine Environmental Research†	Vol. 1 (1978) - Vol. 131 (2017)
Marine Pollution Bulletin†	Vol. 60 (2010) - Vol. 124 (2017)
Regional Studies in Marine Science†	Vol. 1 (2015) - Vol. 15 (2017)
Wetlands	2004 - 2016
Wetlands Ecology and Management	Vol. 1 (1989) - Vol. 24 (2016)

b) All other journals searched as part of CE (245):

* signifies that the journal is of wider relevance to this synopsis.

JOURNAL	Volume/Year searched
Acta Chiropterologica	Vol. 1 (1999) - Vol. 19 (2017)
Acta Herpetologica	Vol. 1 (2006) - Vol. 7 (2012)
*Acta Oecologica-International Journal of Ecology**	Vol. 11 Issue 1 (1990) - Vol. 84 (2017)
Acta Theriologica Sinica	Vol. 20 Issue 1 (2000) - Vol. 37 Issue 4 (2017)
African Bird club Bulletin	2010-2016
African Journal of Ecology	Vol. 1 Issue 1 (1963) - Vol. 54 Issue 4 (2016)
African Journal of Herpetology (formerly The Journal of the Herpetological Association of Africa)	Vol. 38 (1990) - Vol. 61 Issue 1 (2012)
African Primates	1995 - 2012
African Zoology	Vol. 1 (1979) - Vol. 48 (2013)
Agriculture, Ecosystems and Environment	Vol. 10 Issue: 3 (1983) - Vol. 250 (2017)
Agroforestry Systems (Springer)	Vol. 1 (1982) - Vol. 71 (2007)
*Aliens: The Invasive Species Bulletin (IUCN)**	Vol. 1 (1995) - Vol. 33 (2013)
*Ambio**	Vol. 1 Issue 1 (1972) - Vol. 40 Issue 1 (2011)
American Journal of Primatology	1981-2014
American Naturalist	Vol. 1 Issue 1 (1867) - Vol. 190 (2017)
Amphibian and Reptile Conservation	Vol. 1 (1996) - Vol. 9 (2016)
Amphibia-Reptilia	Vol. 1 (1980) - Vol. 37 (2016)
*Animal Biology**	Vol. 53 Issue 1 (2003) - Vol. 63 Issue 3(2013)
*Animal Conservation**	Vol. 1 (1998) - Vol 21 Issue 1 (2018)
Animal Welfare	Vol. 1 (1992) - Vol. 25 (2016)
Annales Zoologici Fennici	Vol. 1 (1964) - Vol. 50 Issue 4 (2013)
Annales Zoologici Societatis Zoologicae Botanicae Fennicae Vanamo	Vol. 1 (1932) - Vol. 25 (1963)
*Annual Review of Ecology, Evolution, and Systematics (formerly Annual Review of Ecology and Systematics)**	Vol. 1 (1970) - Vol. 48 (2017)
Anthrozoos	Vol. 1 (1987) - Vol. 26 (2013)
Apidologie	Vol. 1 (1958) - Vol. 40 (2009)
*Applied Animal Behaviour Science**	Vol. 12 Issue 1 (1988) - Vol. 151 (2014)
Applied Herpetology	Vol. 1 (2003) - Vol. 6 (2009)
Applied Vegetation Science	Vol. 1 Issue 1 (1998) - Vol. 20 Issue 4 (2017)
Aquatic Botany	Vol. 1 (1975) - Vol. 137 (2017)
Aquatic Mammals	Vol. 1 (1972) - Vol. 43 (2017)
Arid Land Research and Management (formerly Arid Soil Research and Rehabilitation)	Vol. 1 Issue 1 (1987) - Vol. 27 Issue 4 (2013)
Asian Primates	2008- 2012
Asiatic Herpetological Research	Vol. 5 (1993) - Vol. 11 (2008)
Auk	(1980 - 2016)
Austral Ecology	Vol. 1 (1977) - Vol. 42 (2017)
Australasian Journal of Herpetology	Vol. 1 (2009) - Vol. 15 (2012)
Australasian Plant Conservation	Vol. 1 issue 1 - Vol. 19 issue 2
Australian Mammalogy	Vol. 22 Issue 1 (2000) - Vol. 39 Issue 2 (2017)
Avian Conservation and Ecology	Vol. 1 (2005) - Vol. 11 (2016)
*Basic and Applied Ecology**	Vol. 1 Issue 2 (2000) - Vol. 25 (2017)

*Behavioral Ecology**	Vol. 1 Issue1 (1990) - Vol. 24 Issue 4 (2013)
Behaviour	Vol. 1 Issue 1 (1948)- (2013)
Bibliotheca Herpetologica	1999 - 2012
Biocontrol (formerly Entomophaga)	Vol. 1 Issue 1 (1956) - Vol. 61 Issue 6 (2016)
Biocontrol Science and Technology	Vol.1 issue 1 (1991) - Vol. 6 issue 2 (1996)
*Biodiversity and Conservation**	Vol. 3 Issue 1 (1994) - Vol. 26 Issue 14 (2017)
*Biological Conservation (Elsevier)**	Vol. 21 (1981) - Vol. 216 (2017)
*Biological Control**	Vol. 1 issue 1 (1991) - Vol. 107 (2017)
*Biological Invasions (Springer)**	Vol. 1 (1999) – Vol. 19 Issue 6 (. 2017)
*Biology and Environment**	Vol. 93 (1993) - Vol. 117 (2017)
*Biology Letters**	Vol. 1 Issue 1 (2005) - Vol. 9 Issue 12 (2017)
Biotropica	Vol. 1 (1990) - Vol. 49 (2017)
Bird Conservation International	1991 - 2016
Bird Study	1980 - 2016
Boreal Environment Research	Vol. 1 Issue 1 (1996) - Vol. 19 Issue 1 (2014)
Bulletin of the Herpetological Society of Japan	1999 - 2008
Canadian Field-Naturalist (formerly Ottawa Naturalist)	Vol. 1 Issue 1 (1987) - Vol. 131 Issue 4 (2017)
Canadian Journal of Forest Research	Vol. 1 (1971) - Vol. 43 (2013)
Caribbean Journal of Science	Vol.1 (1961)-Vol.46 Issue 2-3(2013)
Chelonian Conservation and Biology	Vol. 5 (2006) - Vol. 12 (2013)
*Community Ecology**	Vol. 1 (2000) - Vol. 13 (2012)
*Conservation Biology**	Vol. 1 (1987) - Vol. 31 Issue 6 (2017)
*Conservation Evidence**	Vol. 1 (2004) - Vol. 15 (2018)
*Conservation Genetics**	Vol. 1 Issue 1 (2000) - Vol. 14 Issue 4 (2013)
*Conservation Letters**	Vol. 1 Issue 1 (2008) - Vol. 10 Issue 6 (2017)
Contemporary Herpetology	1998 - 2009
Contributions to Primatology	1974 - 1991
Copeia	1910 - 2003 & Vol. 1 (2000) - Vol. 17 (2016)
Cunninghamia	Vol. 1 (1981) - Vol. 16 (2016)
Current Herpetology (formerly Acta Herpetologica japonica, and Japanese Journal of Herpetology)	Vol. 1 (1964) - Vol. 31 (2012)
Dodo	Vol. 14 (1977) - Vol. 37 (2001)
Ecological and Environmental Anthropology	2005 - 2008
*Ecological Applications**	Vol. 1 Issue 1 (1991) - Vol. 27 Issue 8 (2017)
*Ecological Indicators**	2001 - 2007
*Ecological Management and Restoration**	Vol. 1 (2000) - Vol. 18 (2017)
*Ecological Restoration**	Vol. 1 (1981) - Vol. 35 Issue 4 (2017)
*Ecology**	Vol. 17 Issue 1 (1936) - Vol. 97 Issue 12 (2017)
*Ecology Letters**	Vol. 1 Issue 1 (1998) - Vol. 16 issue 9 (2013)
Ecoscience	Vol. 1 Issue1 (1994) - Vol. 20 Issue 2 (2013)
*Ecosystems**	Vol. 1 Issue1 (1998) - Vol. 16 Issue 8 (2013)
Emu	1980 - 2016
*Endangered Species Research**	Vol. 1 (2004) - Vol. 34 (2017)
*Environmental Conservation**	Vol. 1 Issue 1 (1974) - Vol. 44 Issue 4 (2017)
*Environmental Evidence**	Vol. 1 (2012) - Vol. 6 (2017)
*Environmental Management**	Vol. 1 (1977) - Vol. 60 Issue 6 (2017)

Environmentalist	Vol. 1 Issue 1 (1981) - Vol. 8 Issue 1 (1988)
Ethology Ecology & Evolution	Vol. 1 Issue 1 (1989) - Vol. 26 Issue 1 (2014)
European Journal of Soil Science	Vol. 1 (1950) - Vol. 63 (2012)
European Journal of Wildlife Research (Springer) (formerly Zeitschrift für Jagdwissenschaft)	Vol. 1 (1955) - Vol. 63 Issue 6 (2017)
Evolutionary Anthropology	1992 - 2014
*Evolutionary Ecology**	Vol. 1 Issue1 (1987) - Vol. 28 Issue 1 (2014)
*Evolutionary Ecology Research**	Vol. 1 Issue1 (1999) - Vol. 15 Issue 6 (2014)
Fire Ecology	Vol. 1 Issue 1 (2005) - Vol. 12 Issue 1 (2016)
Folia Primatologica	1963 - 2014
Folia zoologica	Vol. 4 (1959) - Vol. 62 (2013)
Forest Ecology and Management	Vol. 1 (1976) - Vol. 294 (2013)
Freshwater Biology	2004 - 2017
Freshwater Science (formerly Journal of the North American Benthological Society)	Vol. 1 issue 1 (1982) - Vol. 36 Issue 3 (2017)
*Functional Ecology**	Vol. 1 Issue 1 (1987) - Vol. 27 Issue 3 (2013)
Genetics and Molecular Research	Vol. 1 Issue 1 (2002) - Vol. 12 Issue 2 (2013)
Geoderma	Vol. 1 (1967) - Vol. 180 (2012)
Gibbon Journal	2005 - 2011
*Global Change Biology**	Vol. 1 Issue 1 (1995) - Vol. 23 Issue 12 (2017)
*Global Ecology and biogeography**	Vol. 1 Issue 6 (1991) - Vol. 23 Issue 2 (2014)
Grass and Forage Science	Vol. 35 Issue 1 (1980) - Vol. 72 Issue 4 (2017)
Herpetofauna	2003 - 2007
Herpetologica	Vol. 1 (1936) - Vol. 68 (2012)
Herpetological Conservation and Biology	Vol. 1 (2006) - Vol. 7 (2012)
Herpetological Journal	Vol. 1 (1985) - Vol. 22 (2002)
Herpetological Monographs	Vol. 1 (1982) - Vol. 26 (2012)
Herpetological Review	1967 - 2014
Herpetology Notes	2008 - 2014
*Human Wildlife Interactions (formerly Human Wildlife Conflicts)**	Vol. 1, Issue 1 (2007) - Vol. 11 Issue 3 (2017)
Hystrix, the Italian Journal of Mammalogy	Vol. 1 Issue1 (1986) - Vol. 28 Issue 2 (2017)
Ibis	1980 - 2016
iForests	Vol. 1 (2008) - Vol. 9 (2016)
*Integrative Zoology**	Vol. 1 Issue 1 (2006) - Vol. 8 Issue 2 (2013)
International Journal of Pest Management (formerly PANS Pest Articles & News Summaries, PANS, and Tropical Pest)	Vol. 1 (1969) - Vol. 25 (1979)
International Journal of Primatology (Springer)	1980 - 2012
International Journal of the Commons	Vol. 1 Issue 1 (2007) - Vol. 10 Issue 2 (2016)
International Journal of Wildland Fire	Vol. 1 Issue 1 (1991) - Vol. 25 Issue 11 (2016)
International Wader Studies	1970 – 1972
International Zoo Yearbook	Vol. 1 (1960) - Vol. 49 (2015)
Invasive Plant Science and Management	Vol. 1 (2008) - Vol. 9 (2016)
Israel Journal of Ecology & Evolution	Vol. 12 Issue 1 (1963) - Vol. 59 Issue 2 (2013)
Italian Journal of Zoology	Vol. 45 Issue 1 (1978) - Vol. 80 Issue 4 (2013)
*Journal for Nature Conservation**	Vol. 10 (2002) - Vol. 40 (2017)

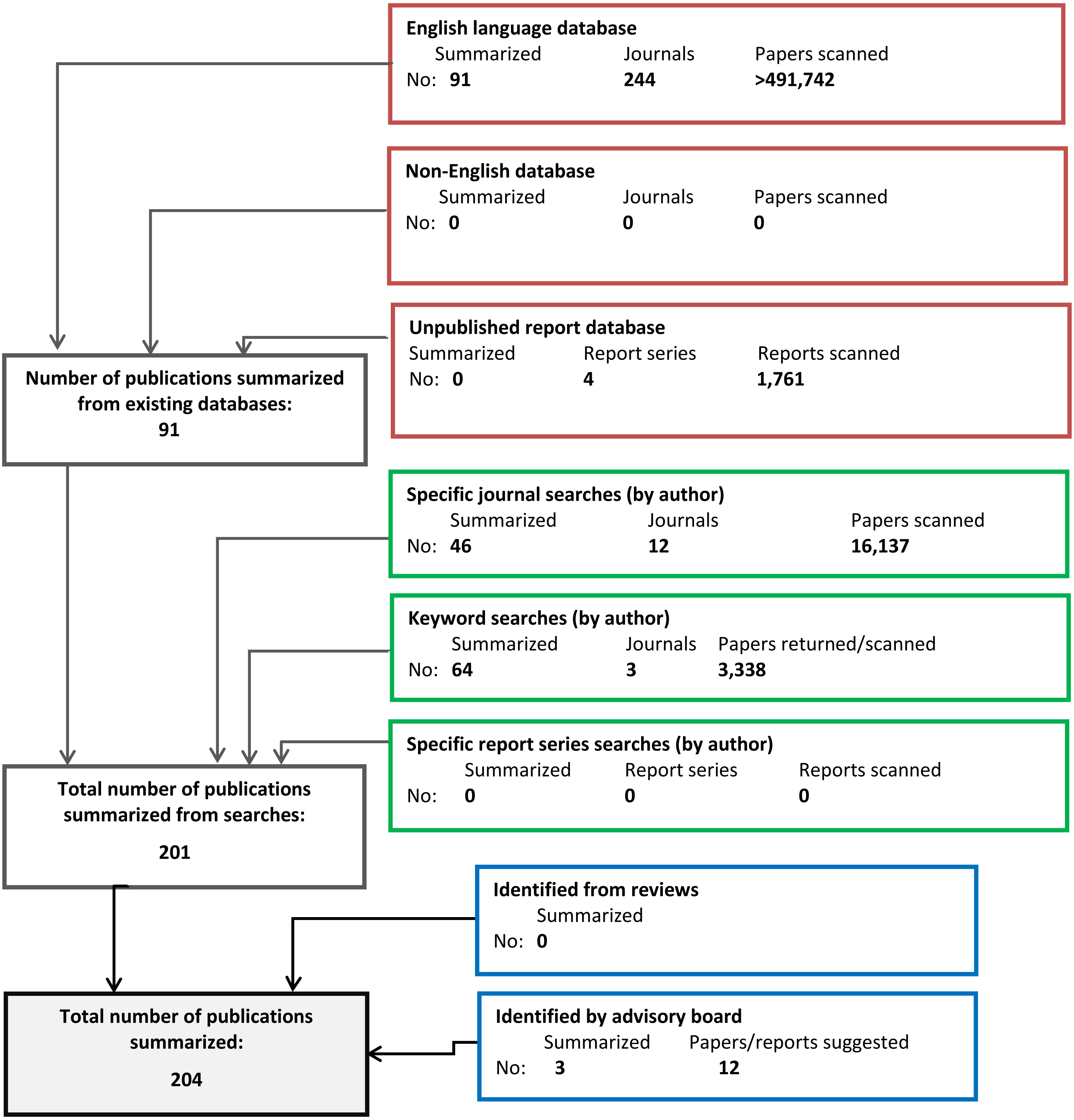
*Journal of Animal Ecology (BES)**	Vol. 1 (1932) - Vol. 86, Issue (2017)
Journal of Apicultural Research	Vol. 1 (1962) - Vol. 48 (2009)
*Journal of Applied Ecology**	Vol. 1, Issue 1 (1964) - Vol. 54 Issue 6 (2017)
Journal of Aquatic Plant Management (formerly Hyacinth Control Journal)	Vol. 1 (1962) - Vol. 54 (2016)
Journal of Arid Environments	Vol. 24 (1993) - Vol. 136 (2017)
Journal of Avian Biology (formerly Ornis Scandinavica)	1980 - 2016
Journal of Cetacean Research and Management	Vol. 1 (1999) - Vol. 12 (2012)
*Journal of Ecology**	Vol. 21 Issue 1 (1933) - Vol. 105 Issue 6 (2017)
*Journal of Environmental Management**	Vol. 1 (1973) - Vol. 204 (2017)
Journal of Field Ornithology	1980 - 2016
Journal of Forest Research	Vol. 1 Issue 1 (1996) - Vol. 22 Issue 1 (2017)
Journal of Great Lakes Research	Vol. 1 (1975) – Vol. 43 (2017)
Journal of Herpetological Medicine and Surgery	2009 - 2013
Journal of Herpetology	1968 - 2015
Journal of Insect Conservation	Vol. 1 (1997) - Vol. 13 (2009)
Journal of Insect Science	Vol. 3 Issue 1 (2003) - Vol. 18 Issue 1 (2018)
Journal of Kansas Herpetology/Collinsorum	2002 - 2014
Journal of Mammalian Evolution	Vol. 1 Issue 1 (1993) - Vol. 21 Issue 1 (2014)
Journal of Mammalogy	Vol. 1 (1919) - Vol. 98 (2017)
Journal of Mountain Science	Vol. 1 Issue 1 (2004) - Vol. 13 Issue 8 (2016)
Journal of Negative Results: Ecology and Evolutionary Biology	Vol. 1 (2004) - Vol. 11 (2016)
Journal of Ornithology	Vol. 145 Issue 1 (2004) - Vol. 159 Issue 1 (2018)
Journal of Primatology	2012 - 2013
Journal of Raptor Research	1966 - 2016
*Journal of Threatened Taxa**	Vol. 1 Issue 1 (2009) - Vol. 5 Issue 2 (2013)
Journal of Tropical Ecology	Vol. 2 (1986) - Vol. 33 (2017)
Journal of Vegetation Science	Vol. 1 Issue 1 (1990) – Vol. 28 Issue 3 (2017)
Journal of Wildlife Diseases	Vol. 1 (1965) - Vol. 48 (2012)
Journal of Wildlife Management	Vol. 9 Issue 4 (1945) - Vol. 81 Issue 8 (2017)
Journal of Zoo & Aquarium Research	Vol. 1 (2013) - Vol. 4 (2016)
*Journal of Zoology**	Vol. 149 (1966) - Vol. 303 Issue 4 (2017)
Jurnal Primatologi Indonesia	2009
Kansas Herpetological Society Newsletter	1977, 1983, 1998, 2001
Lake and Reservoir management	Vol. 1 Issue 1 (1984) - Vol. 32 Issue 4 (2016)
Land degradation and development	Vol. 1 Issue 1 (1989) - Vol. 27 Issue 8 (2016)
Land Use Policy	Vol. 1 (1984) - Vol. 29 (2012)
Latin American Journal of Aquatic Mammals	Vol. 1 (2002) - Vol. 11 (2016)
Lemur News	1993 - 2012
Mammal Research (formerly Acta Theriologica)	Vol. 1 (1977) - Vol. 62 (2017)
Mammal Review	Vol. 1 (1970) - Vol. 47 (2017)
Mammal Study	Vol. 30 Issue 1 (2005) - Vol. 42 Issue 4 (2017)
Mammalia	Vol. 1 (1937) - Vol. 31 (2017)
Mammalian biology	Vol. 67 Issue 1 (2002) - Vol. 87 (2017)
Mammalian Genome	Vol. 1 Issue 1 (1991) - Vol. 24 Issue 8 (2013)

*Management of Biological Invasions**	Vol. 1 (2010) - Vol. 7 (2016)
Marine Mammal Science	Vol. 1 (1985) - Vol. 13 (2017)
Mires and Peat	Vol. 1 (2006) - Vol. 18 (2016)
Natural Areas Journal	Vol. 12 Issue 3 (1992) - Vol. 37 Issue 2 (2017)
NeoBiota	Vol. 9 (2011) - Vol. 34 (2017)
Neotropical Entomology	Vol. 30 (2001) – Vol. 36 (2007)
Neotropical Primates	1993 - 2014
New Journal of Botany	Vol 1, Number 1 (June 2011) - Feb 2013
New Zealand Journal of Zoology	Vol. 1 Issue 1 (1974) - Vol. 44, Issue 4 (2017)
New Zealand Plant Protection	Vol. 53 (2000) – Vol. 69 (2016)
Northwest Science	Vol. 81 Issue 1 (2007) - Vol. 90 Issue 3 (2016)
*Oecologia**	Vol. 3 Issue 3 (1969) - Vol. 185 Issue 4 (2017)
*Oikos**	Vol. 1 Issue 1 (1949) - Vol. 126 Issue 12 (2017)
Ornitologia Neotropical	Vol. 1 (1990) - Vol. 29 (2018)
*Oryx**	Vol. 1 (1950) - Vol. 51 Issue 4 (2017)
Ostrich	1980 - 2016
Pacific Conservation Biology	Vol. 1 Issue 1 (1993) - Vol. 23 Issue 4 (2017)
Pakistan Journal of zoology	Vol. 36 Issue 1 (2004) - Vol. 45 Issue 3 (2013)
Phyllomedusa	Vol. 1 (2002) - Vol. 11 (2012)
Plant Ecology	Vol. 1 (1948) - Vol. 193 (2007)
Plant ecology & diversity (formerly Transactions of the Botanical Society of Edinburgh)	Vol. 1 (2008) - Vol. 5 (2013)
Plant Protection Quarterly	Vol. 23 (2008) – Vol. 31 (2016)
*PLOS**	Vol. 1 (2006) - Vol. 8 (2013)
Polish Journal of Ecology	Vol. 50 Issue 2 (2002) - Vol. 61 Issue 2 (2013)
*Population Ecology**	Vol. 1 Issue 1 (1952) - Vol. 55 Issue 4 (2013)
Preslia	Vol. 45 Issue 1 (1973) - Vol. 89 Issue 4 (2017)
Primate Conservation	1981 - 2014
Primates	Vol. 1 Issue 1 (1957) - Vol. 54 Issue 4 (2013)
Rangeland Ecology & Management (formerly Journal of Range Management)	Vol.1 (1948) - Vol. 69 (2016)
Rangeland Journal	Vol. 1 Issue 1 (1976) - Vol. 38 Issue 5 (2016)
Raptors Conservation	2005 - 2016
*Restoration Ecology**	Vol. 1 (1993) - Vol. 25 (2017)
Revista Chilena de Historia Natural (RCHN)	Vol. 73 (2000) - Vol. 89 (2016)
Revista de Biologia Tropical	Vol. 24 Issue 2 (1976) - Vol. 35 Issue 3 (2013)
River Research and Applications	1987 - 2016
Russian Journal of Ecology	1993 - 2017
Russian Journal of Herpetology	1994 - 2000
Salamandra	Vol. 26 (2000) - Vol. 52 (2016)
Small Ruminant Research	Vol. 1 (1988) - Vol. 156 (2017)
Soil Biology and Biochemistry	1969 - 2012
Soil Use and Management	1985 - 2012
Solvak Raptor Journal	2007 - 2016
South African Journal of Botany	Vol. 1 (1982) - Vol. 108 (2016)
South African Journal of Wildlife Research	Vol. 1 Issue 1 (1971) - Vol. 144 (2014)
South American Journal of Herpetology	Vol. 1 (2006) - Vol. 7 (2012)

Southern Forests	Vol. 70 Issue 1 (2008) - Vol. 75 Issue 4 (2013)
Southwestern Naturalist	Vol. 1 Issue 1 (1956) - Vol. 58 Issue 2 (2013)
*Systematic Reviews Centre for Evidence-Based Conservation**	All reviews published up to December 2017
The Condor	1980 - 2016
The Rangeland Journal	Vol 1, Issue 1 (1976) - Vol. 38 Issue 5 (2016)
*Trends in Ecology and Evolution**	Vol. 1 Issue 1 (1986) - Vol. 32 Issue 12 (2017)
Tropical conservation science	Vol. 1 Issue 1 (2008) - Vol. 7 Issue 1 (2014)
Tropical Ecology	Vol. 1 Issue 1 (1960) - Vol. 55 issue 1 (2014)
Tropical Grasslands	Vol. 1 (1967) - Vol. 44 (2010)
Tropical Zoology	Vol. 1 Issue1 (1988) - Vol. 26 Issue 4 (2013)
Turkish Jounal of Zoology	Vol. 20 Issue 1 (1996) - Vol. 38 Issue 2 (2014)
Vietnamese Journal of Primatology	2007 - 2009
Waterbirds (formerly Colonial Waterbirds)	1983 - 2016
Weed Biology and Management	Vol. 1 Issue 1 (2001) - Vol. 16 Issue 4 (2016)
Weed Research	Vol. 1 (1961) - Vol. 57 (2017)
West African Journal of Applied Ecology	Vol. 1 (2000) - Vol. 24 (2016)
Western North American Naturalist	Vol. 60 Issue 2 (2000) - Vol. 72 Issue 2 (2016)
Wildfowl	Vol. 1 (1948) - Vol. 66 (2016)
Wildlife Biology	Vol. 1 (1995) - Vol. 19 (2013)
Wildlife Monographs	Vol. 1 (1958) - Vol. 183 (2013)
Wildlife Research (CSIRO publishing) (formerly CSIRO Wildlife Research)	Vol. 1 (1956) - Vol. 43 (2016)
Wildlife Society Bulletin	Vol. 1 (1973) - Vol. 41 (2017)
Wilson Journal of Ornithology (formerly Wilson Bulletin)	1980 - 2016
Zhurnal Obshchei Biologii	Vol. 33 Issue 1 (1972) - Vol. 74 Issue 6 (2013)
Zoo Biology	Vol. 1 Issue 1 (1982) - Vol. 35 Issue 2 (2016)
Zookeys	Vol. 1 (2008) - Vol. 312 (2013)
Zoologica Scripta	Vol. 1 Issue 1 (1971) - Vol. 43 Issue 1 (2014)
Zoological Journal of the Linnean Society	Vol. 1 Issue 1 (1856) - Vol. 169 Issue 4 (2013)
Zootaxa	2004 - 2014

Appendix 3. Literature reviewed for the Subtidal Benthic Invertebrate Synopsis

The diagram below shows the number of journals searched for this synopsis, the total number of publications scanned (at title and abstract) within those, and the number of publications that were summarised from each source of literature. In red boxes are the references obtained from the Conservation Evidence database during all the authors searches. In green boxes are the searches undertaken by the authors of this synopsis. In blue boxes are the studies obtained from systematic reviews and as direct suggestions from the Advisory Board.



Appendix 4. Strings used during keyword searches for the Subtidal Benthic Invertebrate Synopsis

Five journals were searched partially (ICES; JMBA) or fully (Estuarine, Coastal and Shelf Science; Fisheries Research; MEPS) using keyword strings for the years 2000-2017. The partially searched journals were later on systematically searched by Conservation Evidence authors using the standard subject-wide evidence synthesis methodology described in the introduction (see also Sutherland et al. in press). The exact strings used for the searches are listed below:

Journal	Search platform	Keyword search	Search on
Marine Ecology Progress Series	Scopus	(SRCTITLE (marine AND ecology AND progress AND series)) AND (benthic AND habitat AND protection) OR (benthic AND habitat AND conservation) OR (benthic AND habitat AND management) OR (benthic AND habitat AND rehabilitation) OR (benthic AND habit at AND restoration) OR (benthic AND habitat AND intervention)	Abstract, title and keywords
Journal of the Marine Biological Association of the UK	Scopus	(SRCTITLE (journal AND of AND the AND marine AND biological AND association AND of AND the AND united AND kingdom)) AND (benthic AND habitat AND protection) OR (benthic AND habitat AND conservation) OR (benthic AND habitat AND management) OR (benthic AND habitat AND rehabilitation) OR (benthic AND habitat AND restoratio n) OR (benthic AND habitat AND intervention)	Abstract, title and keywords
Fisheries Research	Scopus	(SRCTITLE (fisheries AND research)) AND (benthic AND habitat AND protection) OR (benthic AND habitat AND conservation) OR (benthic AND habitat AND management) OR (benthic AND habitat AND rehabilitation) OR (benthic AND habitat AND restoratio n) OR (benthic AND habitat AND intervention)	Abstract, title and keywords
Estuarine, Coastal and Shelf Science	Scopus	(SRCTITLE (estuarine AND coastal AND shelf AND science)) AND (benthic AND habitat AND protection) OR (benthic AND habitat AND conservation) OR (benthic AND habit at AND management) OR (benthic AND habitat AND rehabilitation) OR (benthic AND habitat AND restoration) OR (benthic AND habitat AND intervention)	Abstract, title and keywords
ICES Journal of Marine Science	Scopus	(SRCTITLE (ices AND journal AND of AND marine AND science)) AND (benthic AND ha bitat AND protection) OR (benthic AND habitat AND conservation) OR (benthic AND habitat AND management) OR (benthic AND habitat AND rehabilitation) OR (benthic AND habitat AND restoration) OR (benthic AND habitat AND intervention)	Abstract, title and keywords