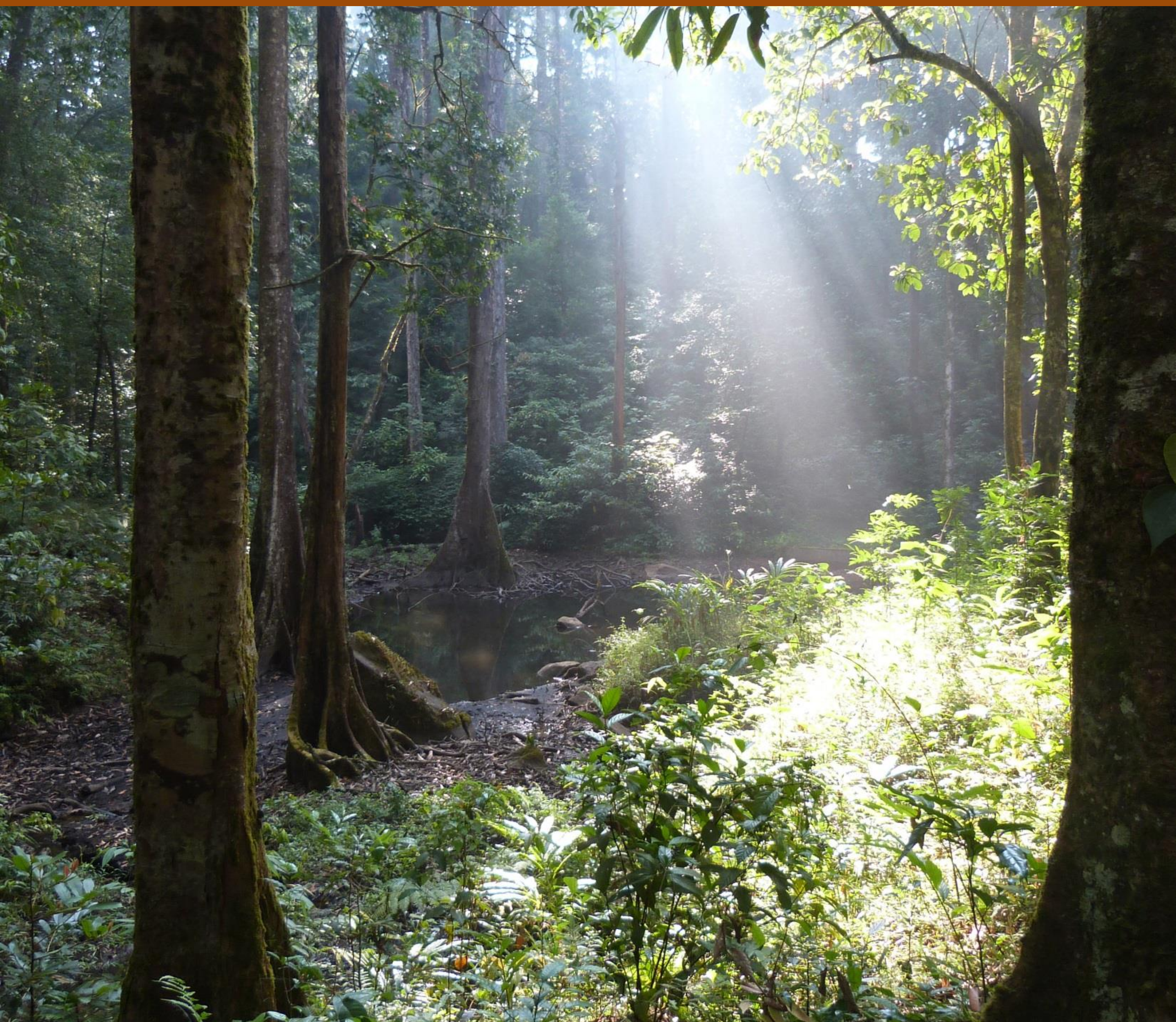
Forest Conservation

Global evidence for the effects of interventions



Har'el Agra, Yohay Carmel, Rebecca K. Smith & Gidi Ne'eman

SYNOPSES OF CONSERVATION EVIDENCE SERIES

Forest Conservation

Global evidence for the effects of interventions

Har’el Agra, Yohay Carmel, Rebecca K Smith, and Gidi Ne’eman

Synopses of Conservation Evidence

Agra, H., Carmel, Y., Smith, R.K. and Ne'eman, G. (2016) *Forest conservation: Global evidence for the effects of interventions*. Synopses of Conservation Evidence Series. University of Cambridge, Cambridge, UK

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:

Prof Ellison Aaron, Harvard Forest and Harvard University, USA
Prof. Jon E. Keeley, USGS Sequoia and Kings Canyon Field Station and UCLA, USA
Prof. Marinus Werger, Utrecht University, The Netherlands
Dr. Le Quoc Huy, Vietnam Academy of Forest Science, Vietnam

About the authors

Har’el Agra is a postdoctoral fellow at the faculty of Civil and Environmental Engineering, Technion- Israel Institute of Technology, Israel.
Yohay Carmel is an associate professor at the Faculty of Civil and Environmental Engineering, The Technion, Haifa.
Rebecca K. Smith is a Senior Research Associate in the Department of Zoology, University of Cambridge, UK.
Gidi Ne'eman is a professor at the Department of Biology and Environment, University of Haifa-Oranim.

Acknowledgements

We would like to thank Dr Lynn Dicks, Professor William Sutherland, Simon Schowanek, Dr Stephanie Prior and Tomer Gueta for their help and advice.

We also thank our funders.

1. About this book

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 4,300 studies so far) on the effects of interventions to conserve biodiversity	• Include evidence on the basic ecology of species or habitats, or threats to them
• List all realistic interventions for the species group or habitat in question, regardless of how much evidence for their effects is available	• Make any attempt to weight or prioritize interventions according to their importance or the size of their effects
• Describe each piece of evidence, including methods, as clearly as possible, allowing readers to assess the quality of evidence	• 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 we have covered the most important literature	• Provide recommendations for conservation problems, but instead provide scientific information to help with decision-making

Who is this synopsis for?

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

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

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

The Conservation Evidence project

The Conservation Evidence project has four parts:

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

2) An ever-expanding **database of summaries** of previously published scientific papers, reports, reviews or systematic reviews that document the effects of interventions.

3) Synopses of the evidence captured in parts one and two on particular species groups or habitats. Synopses bring together the evidence for each possible intervention. They are freely available online and available to purchase in printed book form.

4) ***What Works in Conservation*** is an assessment of the effectiveness of interventions by expert panels, based on the collated evidence for each intervention for each species group or habitat covered by our synopses.

These resources currently comprise over 4,300 pieces of evidence, all available in a searchable database on the website www.conservationevidence.com.

Alongside this project, the Centre for Evidence-Based Conservation (www.cebc.bangor.ac.uk) and the Collaboration for Environmental Evidence (www.environmentalevidence.org) carry out and compile systematic reviews of evidence on the effectiveness of particular conservation interventions. These systematic reviews are included on the Conservation Evidence database.

Of the 114 forests conservation interventions identified in this synopsis, none have previously been the subject of a specific systematic review.

Scope of the forest conservation synopsis

This synopsis covers evidence for the effects of conservation interventions for forests. These are interventions designed to benefit the forest habitat, not specific plant species, or animal populations within forests (those are covered in separate synopses). We included the following types of forests: tropical forests, temperate forests, woodland, scrubland, shrubland and dry forests. We excluded savannahs, wetlands, and mangroves. Evidence from all around the world is included. Any apparent bias towards evidence from some regions reflects the current biases in published research papers available to Conservation Evidence.

Much of the conservation effort in forests concerns forest restoration. Therefore, we added to this synopsis a separate section with actions to support restoration actions, where the major goal is the establishment of tree seedlings planted for restoration. The interventions in most cases are the same interventions as in the main section that describes the response of natural forest components, but the results in this section are the response of planted seedlings.

How we decided which conservation interventions to include

A list of interventions was developed and agreed in partnership with an Advisory Board made up of international conservationists and academics with expertise in forest conservation. We have included actions that have been carried out or advised to support natural forests, including the trees and understory (not animals or specific plant species). Additional interventions were proposed towards the end of this project, to try to include all actions to support natural forests. These interventions were not searched for in the two specialist forest journals or as part of the general keyword search described below. Appendix 1 lists all the interventions classified into three categories: (a) interventions included in all searches and relevant articles found and summarized (n=60), (b) interventions included in all searches but no relevant articles found (n=17), and (c) interventions not included in the search of the two specialist journal or keyword search (n=38).

The list of interventions was organized into categories based on the International Union for the Conservation of Nature (IUCN) classifications of direct threats and conservation actions.

How we reviewed the literature: in addition to evidence already captured by the Conservation Evidence project, we searched the following sources for evidence relating to forests:

- Two specialist forest journals, from their first publication to the end of 2013 (*Forest Ecology and Management, Canadian Journal of Forest Research*), and thirty general conservation journals over the same time period (see: http://conservationevidence.com/site/page?view=methods). These sources yielded 2,150 relevant articles.
- A general keyword search: we used the Scopus engine (www.scopus.com) to search for additional relevant papers. We defined three groups of keywords: aims, forest types and interventions. We searched the whole Scopus database for studies that included at least one keyword in each of these three groups (see Table 1 below). This search found 13,000 papers.

All these articles were then tested for inclusion in *Conservation Evidence*. The criteria for inclusion of studies in the Conservation Evidence database are as follows:

- There must have been an intervention carried out that conservationists would do.
- The effects of the intervention must have been monitored quantitatively.

These criteria exclude studies examining the effects of specific interventions without actually doing them. For example, predictive modelling studies and studies looking at species distributions in areas with long-standing management histories (correlative studies) were excluded. Such studies can suggest that an intervention could be effective, but do not provide direct evidence of a causal relationship between the intervention and the observed biodiversity pattern. Altogether 297 studies satisfied these criteria, and gave rise to a total of 431 synopsis paragraphs.

Table 1. Keywords used for the search algorithm in *Scopus.* Articles were selected if they had at least one keyword in each group: Aims, Forest types, and Intervention. The symbol * enables a match with any suffix of the word.

AIMS	FOREST TYPES	INTERVENTION	INTERVENTION	INTERVENTION
Conservation	Forest	Application	Grazing	Preparation
Management	Woodland	Burning	Harvesting	Removal
Rehabilitation	Shrubland	Clearcut	Herbicide	Seeding
Restoration	Garrigue	Clipping	Induction	Shading
Revegetation	Mallee	Coppicing	Injection	Spacing
Treatment	Maquis	Cutting	Irrigation	Spraying
	Matorral	Exclusion	Logging	Supply
	Scrub*	Felling	Pest control	Thinning
	Chaparral	Fertiliz*	Pesticide	Weeding
	Fynbos	Fire	Planting	

How the evidence is summarized

Conservation interventions are grouped primarily according to the relevant direct threats, as defined in the IUCN Unified Classification of Direct Threats (http://www.iucnredlist.org/technical-documents/classification-schemes/threats-classification-scheme). In most cases, it is clear which main threat a particular intervention is meant to alleviate or counteract.

Not all IUCN threat types are included, only those that threaten forests, and for which realistic conservation interventions have been suggested.

In two cases where the number of studies for an intervention was more than 50, we separated the intervention into sections according to different response groups. These interventions are: 'thin trees within forests' (separated to 'mature trees', 'young trees', 'understory' and 'non-vascular plants') and 'use prescribed fire' (separated to 'mature trees', 'young trees' and 'understory')

Some important interventions can be used in response to many different threats, and it would not make sense to split studies up depending on the specific threat they were studying. We have therefore separated out these interventions, following the IUCN's Classification of Conservation Actions (http://www.iucnredlist.org/technical-documents/classification-schemes/conservation-actions-classification-scheme-ver2). The actions we have separated out are: 'Habitat protection', 'Habitat restoration and creation', 'Species management' and 'Education and awareness raising'. These respectively match the following IUCN categories: 'Land/water protection', 'Land/water management – Habitat and natural process restoration', 'Species Management' and 'Education and awareness'.

Normally, no intervention or piece of evidence is listed in more than one place, and when there is ambiguity about where a particular intervention should fall there is clear cross-referencing. Some studies describe the effects of multiple interventions. Where a study has not separated out the effects of different interventions, we defined a combined intervention. In the text of each section, studies are presented in chronological order, so the most recent evidence is presented at the end. The summary text at the start of each section groups studies according to their findings.

Background information is provided where we feel recent knowledge is required to interpret the evidence. This is presented separately and relevant references included in the reference list at the end of each background section.

The information in this synopsis is available in two ways (and is likely to be produced as a book soon):

- As a pdf to download from www.conservationevidence.com
- As text for individual interventions on the searchable database at www.conservationevidence.com.

Terminology used to describe evidence

Unlike systematic reviews of particular conservation questions, we do not quantitatively assess the evidence or weight it according to quality within synopses. However, to allow you to interpret evidence, we make the size and design of each trial we report clear. The table below defines the terms that we have used to do this.

The strongest evidence comes from randomized, replicated, controlled trials with paired-sites and before and after monitoring.

Term	Meaning
Site comparison	A study that considers the effects of interventions by comparing sites that have historically had different interventions or levels of intervention.
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 10 replicates is a reasonable amount of replication, although more would be preferable. We provide the number of replicates wherever possible, and describe a replicated trial as 'small' if the number of replicates is small relative to similar studies of its kind. In the case of translocations or release of animals, replicates should be sites, not individuals.
Controlled	Individuals or sites treated with the intervention are compared with control individuals or sites not treated with the intervention.
Paired sites	Sites are considered in pairs, when one was treated with the intervention and the other was not. Pairs of sites are selected with similar environmental conditions, such as soil type or surrounding landscape. This approach aims to reduce environmental variation and make it easier to detect a true effect of the intervention.

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.
Before-and-after trial	Monitoring of effects was carried out before and after the intervention was imposed.
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	If none of the above apply, for example a study looking at the number of people that were engaged in an awareness raising project.

Taxonomy

Taxonomy has not been updated but has followed that used in the original paper. Where possible, common names and Latin names are both given the first time each species is mentioned within each synopsis.

Significant results

Throughout the synopsis we have quoted results from papers. Unless specifically stated, these results reflect statistical tests performed on the results.

Multiple interventions

Some studies investigated several interventions at once. When the effects of different interventions are separated, then the results are discussed separately in the relevant sections. However, often the effects of multiple interventions cannot be separated. When this is the case we defined an intervention that combines two or more treatments (e.g. thinning followed by prescribed burning).

How you can help to change conservation practice

If you know of evidence relating to forest conservation 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.

2. Threat: Residential and commercial development

The biggest threat to biodiversity from residential and commercial development is from the destruction of forested areas that accompanies development. Interventions in response to this threat are described in 'Habitat restoration and creation'. See also 'Threat: Pollution' and 'Threat: Transportation and service corridors'.

Key messages – housing and urban areas

Compensate for woodland removal with compensatory planting

We found no evidence for the effects of compensatory planting in new development areas on forests.

Incorporate existing trees or woods into the landscape of new developments

We found no evidence for the effects of incorporating existing trees or woods into the landscape of new developments on forests.

Provide legal protection of forests from development

We found no evidence for the effects of providing legal protection of forests from development.

Key messages – tourism and recreation areas

Create managed paths/signs to contain distrbance

We found no evidence for the effects of creating managed paths or signs to contain disturbance on forests.

Re-route paths, control access or close paths

We found no evidence for the effects of re-routing paths, controlling access or closing paths on forests.

Use warning signs to prevent fire

We found no evidence for the effects of using warning signs to prevent fire on forests.

Adopt ecotourism

We found no evidence for the effects of adopting ecotorism on forests.

Housing and urban areas

2.1. Compensate for woodland removal with compensatory planting

- We found no evidence for the effects of compensatory planting on forests.

Background

Forests or woods that are converted into new development areas suffer from almost total loss of the original biodiversity. Therefore, carrying out compensatory planting in new development areas should have a positive effect on biodiversity. However, the species assemblages are likely to be very different to those of the original forest. Using the dominant trees and shrub species of the original forest for the compensatory planting may contribute to the conservation of some of the original forest species.

2.2. Incorporate existing trees or woods into the landscape of new developments

- We found no evidence for the effects of incorporating existing trees or woods into the landscape of new developments on forests.

Background

Forests or woods that are converted into new development areas suffer from almost total loss of the original biodiversity. Therefore, incorporation of patches of the original vegetation into new developments should have a positive effect on biodiversity. However, for many of the original species, the area of their new 'island habitat' may be insufficient to maintain populations.

2.3. Provide legal protection of forests from development

- We found no evidence for the effects of providing legal protection of forests from development.

Tourism and recreation areas

2.4. Create managed paths/signs to contain disturbance

- We found no evidence for the effects of creating managed paths/signs to contain disturbance on forests.

Background

People can be directed away from vulnerable areas in forests by creating managed routes and placing signs to direct or prohibit access for example.

Studies looking at controlling access are also discussed in "Re-route paths, control access and close paths".

2.5. Re-route paths, control access or close paths

- We found no evidence for the effects of re-routing paths, controlling access or closing paths on forests.

Background
Access to vulnerable areas in forests can be controlled by re-routing paths of prohibiting access to roads and paths for example. Studies looking at path and sign creation are discussed in "Create managed paths/signs to contain disturbance".

2.6. Use signs to prevent fires

- We found no evidence for the effects of using signs to prevent fires on forests.

Background
Using signs may make visitors aware of the fire risk in forests and hence reduce the chances of forest fires.

2.7. Adopt ecotourism

- We found no evidence for the effects of adopting ecotourism on forests.

Background
Ecotourism can be defined as responsible travel to natural areas that aims to conserve the environment, sustain the well-being of the local people, and often involves interpretation and education. Ecotourism in forested areas can therefore potentially benefit the habitat.

3. Threat: Agriculture

Many threats to forests and their biological diversity originate from agriculture, including conversion to agricultural land, loss of agro-forestry systems, overgrazing and unmitigated shifting cultivation. More than 50% of the original temperate forest cover and 95% of tropical, dry forests in Central America has been converted to agriculture (Secretariat of the Convention on Biological Diversity 2002). Agriculture includes many types of land uses such as greenhouses and intensive plantations that cause total extinction of the original forest biodiversity. However, here we focus on livestock farming, which can be much less harmful and may coexist with much of the original forest biodiversity.

Secretariat of the Convention on Biological Diversity (2002) *Review of the status and trends of, and major threats to, the forest biological diversity*. Montreal, SCBD, 164p. (CBD Technical Series no. 7).

Key messages – livestock farming

Use wire fences within grazing areas to exclude livestock from specific forest sections.

Three of four studies, including one replicated, randomized, controlled study in Kenya, Israel, Mexico and Panama found that excluding livestock using wire fences increased the size, density or number of regenerating trees. One study found no effect on tree size and decreased tree density. Four of eight studies, including two replicated, randomized, controlled studies across the world found that excluding livestock using increased biomass, species richness, density or cover of understory plants. Four studies found mixed or no effects on understory plants.

Prevent livestock grazing in forests

One site comparison study in Israel found that preventing cattle grazing increased the density of seedlings and saplings. Two of three studies, including one replicated, controlled study, in Brazil, Costa Rica and the UK found that preventing livestock grazing increased survival, species richness or diversity of understory plants. One study found mixed effects.

Reduce the intensity of livestock grazing in forests

Two studies, including one replicated, randomized, controlled study, in the UK and Greece found that reducing grazing intensity increased the number of tree saplings or understory total weight.

Shorten livestock grazing period or control grazing season in forests

One of two studies, including one replicated, controlled study, in Spain and Australia found that shortening the grazing period increased the abundance and size of regenerating trees. One found no effect native plant species richness. One replicated study in the UK found that numbers of tree seedlings were higher following summer compared to winter grazing.

Provide financial incentives not to graze
We found no evidence for the effects of providing financial incentives not to graze on forests.

Livestock farming

3.1. Use wire fences to exclude livestock from specific forest sections

- Four of eight studies (including two replicated, randomized, controlled studies) in Argentina[5], Australia[11], Belgium[12], Israel[7], New Zealand[10], Spain[4], West Africa[6] and the USA[2] found that excluding livestock using wire fences increased **biomass[4] , species richness[11], density[2] and cover[12] of understory plants**. The other four studies[5,6,7,10] found mixed effects or no effect of livestock exclusion on understory plants.
- Three of four studies (including one replicated, randomized, controlled study) in Mexico[3], Kenya[1], Israel[9] and Panama[8] found that excluding livestock using wire fences increased the size and **density of regenerating trees**[3,9] and the number of regenerating trees[8]. One study[1] found livestock exclusion decreased tree density but not tree size.

Background
Livestock grazing changes habitats, mainly by changing soil properties, plant composition, structure and diversity (Alkemade *et al.* 2013). High grazing pressure can degrade understory species diversity. This is mainly due to decreasing the abundance of palatable herbaceous and low woody species. Using wire fences to excluded livestock from regularly grazed forested areas may increase species diversity (Crawley 1983). Other studies aimed to reduce the detrimental effects of grazing are discussed in 'Prevent livestock grazing in forests'.

Alkemade, R., Reid, R. S., van den Berg, M., de Leeuw, J., Jeuken, M. (2013) Assessing the impacts of livestock production on biodiversity in rangeland ecosystems. *Proceedings of the National Academy of Sciences*, 110, 20900-20905.
Crawley, M.J. (1983) *Herbivory: The dynamics of plant-animal interactions.* Blackwell Scientific Publications, Oxford, UK.

A controlled study in 1986-1990 in tropical dry woodland in Kenya (1) found that **excluding goat grazing decreased the density of** umbrella thorn *Acacia tortilis* **trees but did not affect their height**. The density of umbrella thorn trees was lower in the fenced than in the grazed transects (377 vs 512 trees/ha respectively), while their height was similar (2.2 and 2.5 m respectively). Umbrella thorn trees were monitored in 1990 in five wire fenced (fenced in July 1986 to exclude goats) and five grazed 200 x 20 m transects, in an area with a long history of goat and other livestock grazing.

A replicated, randomized, controlled study in 1999-2001 in temperate coniferous forest in Oregon USA (2) **found that cattle exclusion following cutting of** western juniper *Juniperus occidentalis* **trees** in 1998, **increased seed production of perennial grasses, but did not affect herbaceous plant cover**.

Seed production of perennial grasses was lower in grazed plots (32 kg/ha) than in ungrazed plots (42 kg/ha), while seed production of Sandberg's bluegrass *Poa sandbergii* was similar (5 kg/ha in both). Herbaceous plant cover was similar in grazed and ungrazed plots (16% in both). In 2001, seed production was estimated in five 9 m^2 plots and herbaceous cover was estimated in 0.2 m^2 plots in four pairs of grazed (0.86 cattle/ha for 4-5 days in early 1999 and 2000) and ungrazed plots (0.45 ha).

A replicated, randomized, controlled study in 1997-2001 in tropical dry forest in Mexico (3) found that **cattle exclusion increased trees density and diversity but not species richness**. The total number of woody trees taller than 1.3 m was higher in fenced (650) than in grazed plots (450), the same was true for species diversity (Shannon's index fenced: 1.23; grazed: 0.60). The total number of species was similar between treatments (fenced: 44; grazed: 39). Data were collected in 2001 in eight fenced (with barbwire in 1997) and eight grazed plots (12 × 12 m).

A paired sites, before-and-after trial in 1995-2001 in a Mediterranean Black pine *Pinus nigra* forest in the Pyrenees, Spain (4) found that **grazing exclusion increased the biomass of herbaceous plants and shrubs**. Six years after treatment herbaceous plant and shrub biomasses (kg dry matter/ha) had increased in fenced areas (herbaceous plant: 501 to 1,730; shrub: 1,902 to 5,073) but not in grazed areas (herbaceous plant: 417 to 679; shrub: 1,120 to 1,207). At the beginning of the study herbaceous plant and shrub biomasses were similar in the grazed and fenced areas while six years after both parameters were higher in fenced areas. In 1995, a 10 × 10 m area was fenced to exclude grazing in each of four sites (0.2 cows/ha) each spring and autumn throughout the experiment. Biomass was measured within and outside the fenced area at the end of the grazing season in 1995 and 2001.

A replicated, controlled study in 2002-2006 in temperate mixed forest in Argentina (5) found **no effect of excluding cattle grazing after wildfire on plant species richness and cover**. The total cover of plants was 124% in grazed and 126% in the exclusion plots. Average plant species richness was 32 species/2 m^2 in grazed and 27 species/2 m^2 in fenced plots. Four plots were fenced to exclude cattle and other large herbivores and four unfenced 25 ×25 m plots were installed in March 2002 in an area that was burned by wildfire in 1999. Monitoring was in 2006 in twenty 2 m^2 subplots in each plot.

A replicated, randomized, controlled study in 1993-2003 in savanna woodland in West Africa (6) found **no effect of grazing exclusion on herbaceous plant richness or diversity**. The number of species/0.25 ha (grazed: 14-16; fenced: 13-15) and species diversity (Shannon's index control: 2.7-3.0; exclusion: 2.5-2.8) was similar between treatments. Data were collected in 2003 in four grazed and four fenced (wire fenced to exclude livestock in 1993) treatment plots (0.25 ha) replicated in four blocks, at each of two sites (18 ha).

A replicated, controlled study in 2005-2007 in Mediterranean-type shrubland in Israel (7) **found no effect of cattle exclusion on herbaceous plant species richness**. The number of herbaceous species/plot was similar between grazed and fenced under tree canopies (grazed: 19; fenced: 17) and in open areas (grazed: 82; fenced: 78). In April 2007 herbaceous species were monitored under tree canopies and in open areas in five plots where grazing had been excluded using wire fences (during December 2005) and five grazed plots (0.1 ha; 0.3 cows/ha).

A replicated, controlled study in 2002-2005 in dry tropical forest in Panama (8) found that **cattle exclusion increased basal area, density and species richness of new regenerating trees.** Fenced plots had larger basal area (fenced: 0.03; grazed: 0.01 m^2/plot), density (fenced: 19; grazed: 10 stems/plot) and species richness (fenced: 6; grazed: 4 species/plot) of tree regenerations >1 m height compared to grazed plots. Data were collected in 2005 in 48 grazed (0.6–0.8 head/ha) and forty eight 100 m^2 plots fenced in 2002.

A replicated, controlled, paired sites study in 2006-2009 in Mediterranean-type shrubland in Israel (9) found that **excluding cattle grazing increased the size of regenerating trees after clear cutting.** Three years after clear cutting the average height and diameter of regenerating hawthorn *Crataegus aronia*, terebinth *Pistacia palaestina*, Boissier oak *Quercus boissieri* and Palestine oak *Q. calliprinos* trees were higher in fenced (height: 210; diameter:230 cm) than in grazed plots (height: 70; diameter:110 cm). In 2006, all trees were clearcut in five pairs of grazed (exposed to grazing livestock, 0.3 cows/ha) and fenced (wire fenced in 2005) plots (0.1 ha). Trees were measured in 2009.

A controlled study in temperate mixed forest in New Zealand (10) found that **grazing exclusion decreased exotic plant species richness but did not affect total plant species richness**. The number of exotic plant species/plot was higher in grazed (6.1) and in plots that were ungrazed for 2-10 years (3.8) than in plots that were ungrazed for 10-20 (0.1) or >20 years (0.4). The numbers of native plant species/plot (34, 35, 37 and 34 for grazed, 2-10, 10-20 and >20 years fenced respectively) and total plant species/plot (40, 38, 37 and 35 for grazed, 2-10, 10-20 and >20 years fenced respectively) were similar among treatments. Plants were monitored in 400 m^2 plots in forest fragments: 13 grazed, 10 fenced for 2-10 years, nine fenced for 10-20 years and nine fenced >20 years to exclude cattle and sheep grazing.

A replicated, controlled study in 1981-2010 in Mulga *Acacia aneura* dry forest in Queensland, Australia (11) found that **exclusion of sheep and cattle increased annual grass species richness**. Annual grass species richness was higher in fenced (3.5 species/plot) than in grazed plots (2.6). Species richness was similar between treatments for: all plants (fenced: 18.6; grazed: 15.9), perennial grasses (fenced: 3.3; grazed: .3.3), annual herbaceous plants (fenced: 6.2; grazed: 5.4) and perennial herbaceous plants (fenced: 4.5; grazed: 3.2). In 1981-1983 two treatment plots were established (50 × 50 m): grazed and wire fenced to exclude sheep and cattle, but not kangaroos or rabbits were replicated at three sites regularly grazed by cattle and sheep at 0.1-0.9 dry sheep equivalents/ha. Plant species richness was determined in 2008 in twenty 2 × 7 m plots in each treatment.

A replicated, before-and-after study in 1996-2008 in temperate forest in Belgium (12) found that **excluding cattle grazing increased bramble** *Rubus sp.* **cover and that of some other ground forest plant species**. Bramble cover decreased by 30% in grazed plots and increased by 19% in ungrazed plots. In grazed plots frequencies of English ivy *Hedera helix* and common periwinkle *Vinca minor* decreased (30 vs 0%) (9 vs 0%) respectively, while the cover of oxlip *Primula elatior* remained similar (13 vs 12%). In ungrazed plots frequencies did not change for ivy (26 vs 24%), common periwinkle (5 vs 7%)

and oxlip (16 vs 13%). Percentage cover and the abundance of wood anemone *Anemone nemorosa* were higher in ungrazed than in grazed plots (36 vs 22% cover, 230 vs 100 flowers/ plot respectively). Bramble cover data were collected in 1998 and in 2008 in four plots (20 × 40 m) each divided to equal grazed and ungrazed subplots. Presence/absence of *ivy, common periwinkle and oxlip* (in 2002 and 2008), and cover and frequency of wood anemone (in 2008) were monitored in 206 grazed and 206-225 ungrazed 2 × 2 m plots. Grazing (0.25 cows/ha) began in 2004.

(1) Oba, G. (1998) Effects of excluding goat herbivory on Acacia tortilis woodland around pastoralist settlements in northwest Kenya. *Acta Oecologica*, 19, 395-404.

(2) Bates, J.D. (2005) Herbaceous response to cattle grazing following juniper cutting in Oregon. *Rangeland Ecology and Management*, 58, 225-233

(3) Montero-Solís, F.M., Sánchez-Velásquez, L.R., Pineda-López, M.D.R. Martínez-Rivera, L.M., Moermond, T., and Aguirre, J.C. (2006) S Livestock impact on dynamic and structure of tropical dry forest of the Sierra de Manantlán. Mexico, Journal of Food, *Agriculture and Environment*, 4, 266-270.

(4) Casasús, I., Bernués, A., Sanz, A., Villalba, D., Riedel, J.L., and Revilla, R. (2007) Vegetation dynamics in Mediterranean forest pastures as affected by beef cattle grazing, *Agriculture. Ecosystems & Environment*, 121, 365-370.

(5) Blackhall, M., Raffaele, E., and Veblen, T.T. (2008) Cattle affect early post-fire regeneration in a Nothofagus dombeyi-Austrocedrus chilensis mixed forest in northern Patagonia, Argentina. *Biological Conservation*, 141, 2251-2261.

(6) Savadogo, P., Tiveau, D., Sawadogo, L., and Tigabu, M. (2008) Herbaceous species responses to long-term effects of prescribed fire. grazing and selective tree cutting in the savanna-woodlands of West Africa. *Perspectives in Plant Ecology, Evolution and Systematics*, 10, 179-195.

(7) Agra, H., Ne'eman, G. (2009) Woody species as landscape modulators: their effect on the herbaceous plants in a Mediterranean maquis. *Plant Ecology*, 205, 165-177.

(8) Griscom, H.P., Griscom, B.W., and Ashton, M.S. (2009) Forest regeneration from pasture in the dry tropics of Panama: Effects of cattle, exotic grass, and forested riparia. *Restoration Ecology*, 17, 117-126.

(9) Agra, H., Ne'eman, G. (2011) Quercus calliprinos regrowth advantage under grazing in Mediterranean maquis and its management implications. *Forest Ecology and Management*, 261, 143-147.

(10) Burns, B.R., Floyd, C.G., Smale, M.C., and Arnold, G.C. (2011) Effects of forest fragment management on vegetation condition and maintenance of canopy composition in a New Zealand pastoral landscape. *Austral Ecology*, 36, 153-166.

(11) Fensham, R.J., Silcock, J.L., and Dwyer, J.M. (2011) Plant species richness responses to grazing protection and degradation history in a low productivity landscape. *Journal of Vegetation Science*, 22, 997-1008.

(12) Van Uytvanck, J., and Hoffmann, M. (2012) Impact of grazing management with large herbivores on forest ground flora and bramble understory. *Acta Oecologica*, 35, 523-532.

3.2. Prevent livestock grazing in forests

- Two of three studies (including one replicated, controlled study) in Brazil[1], UK[3] and Costa Rica[2] found that preventing livestock grazing increased **survival[1], species richness and diversity[2] of understory plants**. One study found mixed effects[3].
- One site comparison study in Israel[4] found that preventing cattle grazing increased the **density of oak seedlings and saplings**.

Background

Livestock grazing changes the habitat indirectly by changing soil properties and directly by the removal of vegetation. High grazing pressure may change plant diversity mainly by decreasing the abundance of palatable herbaceous species. It can also cause degradation of forest understory and reduce tree regeneration. Moderate grazing has less negative effects and may increasing shrubby vegetation height. Removing livestock grazing from forests may increase species diversity (Rook *et al.* 2004).

Other studies that aim to reduce the detrimental effects of grazing are discussed in 'Use wire fences within grazing areas to exclude livestock from specific forest sections'.

Rook, A. J., Dumont, B., Isselstein, J., Osoro, K., WallisDeVries, M. F., Parente, G., & Mills, J. (2004) Matching type of livestock to desired biodiversity outcomes in pastures–a review. *Biological Conservation*, 119, 137-150.

A controlled study in 1978-1984 in dry tropical forest in Brazil (1) found that **preventing cattle grazing decreased mortality of shrubs but not the density of tree seedlings**. The average mortality over two years of five shrub species was higher in grazed (7.7-11.7%) than in ungrazed plots (4.5%). The number of tree seedlings (<0.5 m height) was similar in grazed (1.9-5.2 seedlings/m^2) and ungrazed plots (5.8 seedlings/m^2). Mortality of five selected shrub species: *Lippia microphylla, Croton rhamnifolius, Calliandra depauperata, Cordia leucocephala* and *Bauhinia cheilantha*, was calculated for 1980-1982 and for 1982-1984. Mortality data were collected in 6-10 plots (20 × 5 m) in each of three grazed (0.150, 0.100 and 0.075 cattle/ha) and one ungrazed treatment area (40-100 ha). Density of tree seedlings was determined annually in 1979-1984 in five quadrats (2 x 0.5 m) in each plot.

A site comparison study in 1996 in tropical dry forests in Costa Rica (2) found that **preventing livestock grazing increased plant species richness and diversity**. The grazed site had lower numbers of species (grazed: 56; ungrazed: 84) and species diversity (Shannon's index grazed: 2.9; ungrazed: 3.5). Species richness and diversity were calculated using 150 individual plants identified along each of six 2 m wide transects in each ungrazed (since 1990) and grazed sites (grazed in May-January by 1.0-1.4 cattle/ha since 1991).

A replicated, controlled study in 2002-2003 in temperate broadleaf forest in Northern Ireland, UK (3) found that **preventing domestic animal grazing decreased plant species richness but increased the cover of some dominant species**. The number of species within small plots (4 m^2) was higher in grazed (14.0/4 m^2 plot) than ungrazed plots (10.5/4 m^2 plot), while within large plots (196 m^2) the number of species (32.3-28.7/196 m^2 plot) and species diversity (Shannon's index 1.6-1.4) was similar between treatments. Relative cover was lower in grazed than ungrazed plots for: bramble *Rubus fruticosus* (grazed: 3.9%; ungrazed: 9.6%) and bluebell *Hyacinthoides non-scripta* (grazed: 2.5%; ungrazed: 10.1%). Data were collected in 2002-2003 in one small quadrat (2 × 2 m inside 14 × 14 m plot) in each of 52 grazed (sheep, cattle, horses, goats) and 46 ungrazed sites across Northern Ireland (~14,000 km^2).

A site comparison study in 2002 in subtropical dry forest in Israel (4) found that **preventing cattle grazing increased the density of Tabor oak *Quercus ithaburensis* seedlings and young saplings**. Density of young seedlings (<0.15 m height) and of young saplings (0.2 – 1.0 m height) in the grazed site was lower (seedlings: 4/ha; saplings: 14/ha) than in the ungrazed site (seedlings: 12/ha; saplings: 30/ha). Data were collected in 13 plots (333 m^2) established in an ungrazed site (14 ha) and in 33 similar plots established in a grazed site (0.7 cows/ha; 458 ha).

(1) De Albuquerque, S.G. (1999) Caatinga vegetation dynamics under various grazing intensities by steers in the semi-arid Northeast, Brazil. *Journal of Range Management*, 52, 241-248.
(2) Stern, M., Quesada, M., and Stoner, K.E. (2002) Changes in composition and structure of a tropical dry forest following intermittent cattle grazing. *Revista de Biologia Tropical*, 50, 1021-1034.
(3) McEvoy, P.M., Flexen, M., and McAdam, J.H. (2006) The effects of livestock grazing on ground flora in broadleaf woodlands in Northern Ireland. *Forest Ecology and Management* 225, 39-50.
(4) Dufour-Dror, J.M. (2007) Influence of cattle grazing on the density of oak seedlings and saplings in a Tabor oak forest in Israel. *Acta Oecologica*, 31, 223-228.

3.3. Reduce the intensity of livestock grazing in forests

- One replicated study in the UK[1] found that reducing grazing intensity increased the number of **tree saplings**.
- One replicated, randomized, controlled study in Greece[2] found that reducing grazing intensity increased **understory biomass**.

Background
Complete removal of livestock by fencing tends to promote regeneration, mainly in the early stages after removal. This is due to the removal of the disturbance effect of animals, which commonly provides 'niches' for seedling establishment. Reducing grazing intensity rather than complete removal may allow continued income from livestock.

A replicated study in 1986-1993 in temperate woodland in the UK (1) found that **reducing the intensity of sheep grazing increased the numbers of tree saplings**. The number of saplings/100 m^2 was higher in low-intensity (0.54-0.66) than in high- and medium-intensity grazing plots. Four plots for each grazing intensity: high (2.1-3.8 sheep/ha); medium (1.2-2.0 sheep/ha) or low (0.6-1.2 sheep/ha) were established in 1986. Saplings (>30 cm diameter at breast height) were monitored in 2003 in 20 quadrats (10 × 10 m) within each plot.

A replicated, randomized, controlled study in 1991-2005 in a Mediterranean oak forest in central Macedonia, Greece (2) found that **reducing grazing intensity increased understory plant biomass**. Understory production (kg/ha dry matter) was higher in non- and lightly-grazed (~4,500) than in moderately-grazed (~2,800) and heavily-grazed sites (~1,000). A study area of 2,000 ha was divided into six forest segments, each was divided into three areas with different stocking densities (goats and cattle): heavy (15 animals/ha), moderate (5 animals/ha) and light (0.2 animals/ha). Sixty plots (1 m^2) were randomly placed in every grazing treatment in all stands and protected from grazing at the end of 2004. Similar size plots with grazing close to protected (control) plots were sampled for comparison. Overall understory (herbage and browse) production was measured in 1991 and in September 2005.

(1) Hester, A.J., Mitchell, F.J.G., and Kirby, K.J. (1996) Effects of season and intensity of sheep grazing on tree regeneration in a British upland woodland. *Forest Ecology and Management*, 88, 99-106.
(2) Ainalis, A.B., Platis, P.D., and Meliadis, I.M. (2010) Grazing effects on the sustainability of an oak coppice forest. *Forest Ecology and Management,* 259, 428-432.

3.4. Shorten livestock grazing period or control grazing season in forests

- One replicated, controlled study in Spain[3] found that shortening the livestock grazing period increased the **abundance and size of regenerating oak trees**.
- One paired-sites study in Australia[2] found no effect of shortening the livestock grazing period on **native plant species richness**.
- One replicated study in the UK[1] found that the number of **tree seedlings** was higher following summer compared to winter grazing.

Background
Reducing livestock grazing can be done by reducing the number of animals per area unit or by shortening the grazing period during the year. These two methods may vary in their effects on tree regeneration and forest biodiversity, for example if the livestock is removed in critical seasons of seed production and germination.

A replicated study in 1986-1993 in temperate woodland in the UK (1) found that using **summer instead of winter grazing increased the number of tree seedlings**. The number of seedlings was higher following summer (8-17/100 m^2) compared to following winter grazing (4-6/100 m^2). Six summer (May-October) and six winter grazing (October-May) plots were established in 1986. Seedlings (> 1 year old, <30 cm diameter at breast height) were monitored in 2003 in 20 quadrats (10 × 10 m) within each plot.

A paired-sites study in 2006 in temperate woodland in south-eastern Australia (2) found **no effect of different grazing regimes on native plant species richness**. The number of native plant species/plot was similar between treatments (continuous-grazing: 18; rotational-grazing: 15). Monitoring was in two continuous-grazing (livestock had unrestricted access) and two rotational-grazing (<56 days grazing followed by >21 days with no grazing) plots (1 ha) in each of 12 sites (a total of 48 plots).

A replicated study in Mediterranean open woodland in Spain (3) found that **seasonal grazing increased the abundance and height of oak saplings compared to permanent grazing**. Percentage cover of young oaks (seasonal grazing: 9%; permanent: <1%) and young oak height (seasonal: 80 cm; permanent: 40 cm), and density of young and old oak saplings (seasonal: 100-80 saplings/ha; permanent: 20 saplings/ha) were higher with seasonal than permanent grazing. Two to six sites were located in each of nine permanently grazed areas (grazed throughout the year) and nine areas grazed seasonally December to May. Each area was 20-480 ha and had been grazed in the 10 years before treatment. Monitoring was in four 3 × 3 m plots in each site.

(1) Hester, A.J., Mitchell, F.J.G., and Kirby, K.J. (1996) Effects of season and intensity of sheep grazing on tree regeneration in a British upland woodland. *Forest Ecology and Management*, 88, 99-106.
(2) Dorrough, J., Mcintyre, S., Brown, G., Stol, J., Barrett, G., and Brown, A. (2012) Differential responses of plants, reptiles and birds to grazing management, fertilizer and tree clearing. *Austral Ecology,* 37, 569-582.
(3) Carmona, C.P., Azcárate, F.M., Oteros-Rozas, E., González, J.A., and Peco, B. (2013) Assessing the effects of seasonal grazing on holm oak regeneration: Implications for the conservation of Mediterranean dehesas. *Biological Conservation,* 159, 240-247.

3.5. Provide financial incentives not to graze

- We found no evidence for the effects of providing financial incentives not to graze on forests.

4. Threat: Transport and service corridors

The greatest threats from transport and service corridors tend to be from the destruction of habitat and pollution. Interventions in response to these threats are described in 'Habitat restoration and creation' and 'Threat: Pollution'.

Key messages

Maintain/create habitat corridors

We found no evidence for the effects of maintaining or creating habitat corridors on forests.

4.1. Maintain/create habitat corridors

- We found no evidence for the effects of maintaining or creating habitat corridors on forests.

5. Threat: Biological resource use

In early times humans exploited a small proportion of natural resources, mainly by hunting animals and harvesting plants. However, with ever increasing human populations, their affects on natural resources has become very significant. Today, some continents hardly have any primary forests left. Thus much effort is allocated to preserving remaining natural forests. It is important that management also considers the needs of local populations (Janzen 2013).

Janzen, D. (2013) *Race to save the Tropics: Ecology and economics for a sustainable future*. (ed. R. Goodland). Island press.

Key messages - Thinning and wood harvesting

Thin trees within forests

Eleven of 12 studies, including two replicated, randomized, controlled studies, in Brazil, Canada, and the USA found that thinning trees decreased the density and cover of mature trees and in one case tree species diversity. Five of six studies, including one replicated, controlled, before-and-after study, in Australia, Sweden and the USA found that thinning increased mature tree size, the other found mixed effects. One of three studies, including two replicated controlled studies, in the USA found that thinning reduced the number of trees killed by beetles. Six of 12 studies, including two replicated, randomized, controlled studies, in Japan and the USA found that thinning increased the density of young trees and a study in Peru found it increased the growth rate of young trees. One study found thinning decreased the density and five found mixed or no effect on young trees. One replicated, controlled study in the USA found no effect on the density of oak acorns. Twenty five of 38 studies, including 12 replicated, randomized, controlled studies, across the world found that thinning trees increased the density and cover or species richness and diversity of understory plants. Nine studies found mixed and two no effects, and one found a decrease the abundance of herbaceous species. Three of four studies, including one replicated, randomized, controlled study, in Canada, Finland and Sweden found that thinning decreased epiphytic plant abundance and species richness. Three found mixed effects depending on thinning method and species.

Log/remove trees within forests

Three of seven studies, including two replicated, controlled studies, across the world found that logging trees decreased the density and cover of mature trees. Two found it increased tree density and two found no effect. Four of nine studies, including one replicated, randomized, controlled study, across the world found that logging increased mature tree size or diversity. Four found it decreased tree size or species richness and diversity, and two found no effect on mature tree size or diversity. One replicated, controlled study in Canada found that logging increased mature tree mortality rate. One of two replicated controlled studies in Canada and Costa Rica found that logging increased the density of young trees, the other found mixed effects. Eight of 12 studies, including four replicated, randomized, controlled studies, in India, Australia, Bolivia, Canada and the USA found that logging increased the density and cover or species richness and diversity of understory plants. Two studies found mixed and three found no effect. Two of three studies, including one replicated, paired sites study, in Australia, Norway and Sweden found that logging decreased epiphytic plant abundance and fern fertility. One found mixed effects depending on species.

Remove woody debris after timber harvest

Two studies, including one replicated, randomized, controlled study, in France and the USA found no effect of woody debris removal on cover or species diversity of trees. One of six studies, including two replicated, randomized, controlled studies, in Ethiopia, Spain, Canada and the USA found that woody debris removal increased young tree density. One found that it decreased young tree density and three found mixed or no effect on density or survival. One of six studies, including two replicated, randomized, controlled studies, in the USA and France found that woody debris removal increased understory vegetation cover. Five studies found mixed or no effects on understory vegetation cover or species richness and diversity.

Use shelterwood harvest instead of clearcutting

Three replicated, controlled studies in Sweden and the USA found that shelterwood harvesting increased density of trees or plant diversity, or decreased grass cover compared with clearcutting.

Use partial retention harvesting instead of clearcutting

Three studies, including one replicated, randomized, controlled study, in Canada found that using partial retention harvesting instead of clearcutting decreased the density of young trees.

Use summer instead of winter harvest

One replicated study in the USA found no effect of logging season on plant species richness and diversity.

Adopt continuous cover forestry

We found no evidence for the effects of adopting continuous cover forestry on forests.

Use brash mats during harvesting to avoid soil compaction

We found no evidence for the effect of using brash mats during harvesting to avoid soil compaction.

Key messages – harvest forest products

Sustainable management of non-timber products

We found no evidence for the effects of sustainable management of non-timber products on forests.

Adopt certification

One replicated, site comparison study in Ethiopia found that deforestation risk was lower in certified than uncertified forests. One controlled, before-and-after trial in

Gabon found that, when corrected for logging intensity, although tree damage did not differ, changes in above-ground biomass were smaller in certified than in uncertified forests.

Key messages – firewood

Provide fuel efficient stoves

We found no evidence for the effects of providing fuel efficient stoves on forests.

Provide paraffin stoves

We found no evidence for the effects of providing paraffin stoves on forests.

Thinning and wood harvesting

5.1. Thin trees within forests

Background

Thinning is the removal of trees to control the development or enhance the future condition of a forest, by adjusting its density, structure and species composition.

Studies looking at tree removal with the aim of removing biomass are discussed in 'Logging/tree removal within forest'.

5.1.1. Thin trees within forests: effects on mature trees

- Eleven of 12 studies (including two replicated, randomized, controlled studies) in Brazil[1], Canada[2], and the USA[5,6,7,9,10,11,12,15,16,17] found that thinning trees in forests decreased the **density and cover of trees**[1,5,6,7,9,10,11,12,15,16,17]. One study found no effect[2] of thinning on tree density.
- Five of six studies (including one replicated, controlled, before-and-after study) in Australia[4], Sweden[8] and the USA[3,9,10,11] found that thinning trees in forests increased **tree size**[4,8,9,10,11]. One found mixed effects[3] of thinning on tree size.
- One replicated, controlled study in the USA[7] found that thinning trees in forests decreased **tree species richness and diversity**.
- One replicated, site comparison study in the USA[13] found that thinning reduced the number of conifers killed by beetles. Two replicated, controlled studies in the USA[14,18] found no effect of thinning on bark-beetle caused tree mortality.

A replicated, controlled study in 1984-1985 in dry tropical forest in Ceara state, Brazil (1) found that thinning decreased **woody biomass**. Woody biomass (kg/ha) was the highest in unthinned compared to thinned plots (0% cover-retention: 966; 25% cover-retention: 1,058; 55% cover-retention: 1,003; unthinned: 1,891). Four treatment plots (0.1 ha), three thinned (0%, 25% and 55% woody cover retained) and unthinned (95% woody cover) plots were established in 1984 in each of two sites. Data were collected in May 1985 in a subplot protected from grazing (40×50 m) in each plot.

A replicated, controlled study in 1993-1998 in temperate lodgepole pine *Pinus contorta* forest in British Columbia, Canada (2) found **no effect of lodgepole pine thinning on total tree density**. The number of trees/ha was similar in thinned (3,259) and unthinned plots (7,648) Data were collected in 1998 in only three thinned (targeted to retain 1,000 stems/ha) and three unthinned treatment units (1.8-12.6 ha) established in 1993 in each of three study areas.

A replicated, controlled study in 1952-1995 in temperate coniferous forest in Maine, USA (3) found that **thinning decreased trees volume growth rate (final volume plus harvested volume plus mortality minus initial volume) but not their net growth (discounting mortality)**. Annual gross growth (m^3/ha) was lower in thinned (2.51-4.27) than unthinned plots (4.08). Annual net growth was similar between treatments (unthinned: 1.59; thinned: 2.01-3.4). Two unthinned and 18 thinned (thinned in different time intervals following different procedures for 40 years since 1951) treatment units were established inside a 1,619 ha study area. Data were collected every five years from 1951, in a total of 307 plots (0.8 ha).

A replicated, controlled study in 1992–1998 in silvertop ash *Eucalyptus sieberi* forest in Victoria, Australia (4) found that **thinning increased growth rate of the retaining trees**. Annual basal area increase of retained trees in thinned plots was higher (2.2 m^2/ha) than in unthinned plots (1.2 m^2/ha). Data were collected in 1998 in four replicates of two treatment plots (0.16 ha): unthinned and thinned (retaining 45 of the largest trees/plot). Treatments were established in 1992.

A replicated, controlled study in 2000-2005 in temperate coniferous forest in California, USA (5) found that thinning **decreased tree density, basal area and canopy cover and increased tree height**. For all trees >2.5 cm DBH density (thinned: 429/ha; control: 1,109/ha), basal area (thinned: 41 m^2/ha; unthinned: 56 m^2/ha) and canopy cover (thinned: 58%; unthinned: 75%) were lower in thinned plots, while height was higher in thinned plots (thinned: 23 m; unthinned: 16 m). Data were collected in 2005 in 25 plots (0.04 ha) in each of three thinned (to retain 28-34 m^2/ha basal area in 2001, large trees removed and small trees shredded) and three unthinned treatment units (14-29 ha).

A replicated, paired sites study in 2005 in Mediterranean type woodland in Oregon, USA (6) found that **thinning decreased woody canopy cover**. Cover of trees and shrubs >0.3 m tall was higher in unthinned (97%) than thinned (25%) transects. Data were collected in 2005 using 30 pairs of thinned (for fuel reduction between May 1998 and June 2001) and unthinned transects (50 m). Tree cover was measured in five plots (3 m^2) along each transect.

A replicated, controlled study in 2000-2003 in temperate mixed forest in Georgia, USA (7) found that **mechanical thinning decreased tree density and diversity**. In thinned plots the following were lower than in unthinned plots: number of trees/ha (thinned: 212; unthinned; 793), the number of tree species/100 m^2 (thinned: 4.0; unthinned: 8.7) and diversity of trees (Shannon's index in 100 m^2; thinned: 0.78; unthinned: 1.13). Four blocks were established in 2000, each containing thinned (mulching of all broadleaf trees regardless of size, and all pines <20 cm diameter at breast height) and unthinned treatment plots

(110 × 110 m). Data were collected in 2002-2003 in five subplots (10 × 10 m) within each treatment plot.

A replicated, controlled, before-and-after study in 2000-2006 in temperate broadleaf forest in Sweden (8) found that **thinning trees increased oak *Quercus spp.* regrowth rate**. Relative basal area increase of oak trees, i.e. increase/initial area (cm^2/cm^2), was higher in thinned (3.8%) than in unthinned plots (3.2%). Oak tree basal area increase was 106 cm^2 in thinned and 81 cm^2 in unthinned plots. Data were collected before (2002) and after treatment (2006) in 25 pairs of thinned (25% of basal area cut in winter 2002-2003) and unthinned 1 ha plots.

A replicated, controlled study in 2002-2006 in temperate coniferous forest in Washington State, USA (9) found that thinning **decreased tree density and basal area and increased their average stem diameter and canopy height.** Number of trees/ha (thinned: 205; unthinned: 530) and tree basal area (thinned: 17 m^2/ha; unthinned: 34 m^2/ha) were lower in thinned than in unthinned plots. In contrast, the average diameter of trees (thinned: 36 cm; unthinned: 30 cm) and height of the base of the canopy (thinned: 9 m; unthinned: 5 m) were higher in thinned plots. Six thinned (retaining 10–14 m^2/ha basal area) and six unthinned treatment plots (10 ha) were established in 2002-2003. Data were collected 2-4 years after thinning in six 20 × 50 m plots within each treatment unit.

A before-and-after trial in 2003-2006 in temperate coniferous forest in California, USA (10) found that **mechanical thinning decreased tree density and cover, and increased their diameter and canopy height**. Number of trees/ha (before: 427-1,201; after: 183-587) and canopy cover (before: 38%-71%; after: 28%-60%) decreased after thinning. In contrast, the height of the base of the canopy (before: 1.2-4.0 m; after: 3.4-7.6 m) and average diameter (before: 18-38 cm; after: 25-46 cm) increased. Data were collected in 40 plots (0.1 ha) before (2003) and after (2006) thinning followed by tree residue removal. Thinning was carried out in 2003.

A replicated, paired-sites study in 2001-2008 in temperate coniferous forest in Colorado USA (11) found that **thinning trees decreased the number of trees and the bulk density of the canopy, and increased canopy height**. The density of trees >10 cm diameter at breast height (trees/ha) was higher in unthinned (1,691-580) than in thinned plots (383-55). Canopy bulk density of the trees in lodgepole pine *Pinus contorta*, ponderosa pine *P. ponderosa* and mixed conifer forests was greater in unthinned (0.15, 0.12 and 0.14 kg/m^3 respectively) than in thinned plots (0.04, 0.04 and 0.01 kg/m^3 respectively). However, canopy density was similar between thinned and unthinned in pinyon pine/juniper *Pinus edulis/Juniperus sp.* forests (0.02 vs 0.007 kg/m^3). The height of the base of the canopy of the trees in lodgepole pine and mixed conifer forests was higher in thinned (7.7 and 5.1 m respectively) than in unthinned (5.8 and 2.5 m respectively). However canopy height was similar between thinned and unthinned in ponderosa pine (5.4 vs 2.3 m) and pinyon pine/juniper forests (3.9 vs 3.3 m). Trees were measured in 2007-2008 in three 50 m transect in each thinned (mulched with Hydroax© or Morbark© chipper in 2001-2006) and unthinned plot. Plots were replicated within five lodgepole pine, four ponderosa pine, six pinyon pine/juniper and three mixed conifer forests.

A replicated, randomized, controlled study in 1998-2005 in boreal forest in south eastern Alaska, USA (12) **found that thinning decreased canopy cover of conifers.** Canopy cover of conifers was similarly lower in all thinning treatments (50-67%) than in unthinned plots (95%). Two 0.2 ha plots of each of four conifer thinning treatments (retaining 250, 370, 500, and 750 trees/ha) and unthinned plots were replicated in seven 16-18 year old forest sections. Treatments were applied in 1999, data were collected in 2005.

A replicated, site comparison study in 2001-2007 in temperate coniferous forest in California USA (13) found that **thinning reduced the number of conifers killed by fir engraver beetles** *Scolytus ventralis*. The density of ponderosa pine *Pinus ponderosa* and white fir *Abies concolo* trees (>10.2 cm) killed from 2001 to 2007 was lower in commercially thinned (23.8 trees/ha) and salvage-thinned plots (16.4 trees/ha) than in unthinned forest units (44.5 trees/ha). Monitoring was carried out using 20 clusters of four 20 × 100 m transects, in commercially thinned (residual basal area ~37 m^2/ha), salvage-thinned (salvage harvesting of bark beetle-killed trees and live-tree thinning to reduce basal area to ~25 m^2/ha) and in unthinned forest units. Thinning occurred between 1988 and 1998.

A replicated, controlled study in 1998-2003 in temperate coniferous forest in California, USA (14) found **no effect of thinning on tree mortality caused by bark-beetle**. The cumulative percentage of trees killed by bark beetles was similar in thinned (1%) and unthinned plots (3%). All trees killed by bark beetles were recorded in six treatment units (10 ha): three unthinned and three thinned (thinning from below and selection harvest leaving all stems <76.2 cm diameter at breast height and all sugar pine *Pinus lambertiana*, incense cedar *Calocedrus decurrens* and ponderosa pine *Pinus ponderosa*). Thinning was in 1998-1999. Data were collected in 2003.

A replicated, controlled study in 2001-2005 in temperate coniferous forest in Montana, USA (15) found that **thinning decreased trees density and basal area**. Density (thinned: 157 trees/ha; unthinned: 400 trees/ha) and basal area (thinned: 12 m^2/ha; unthinned: 25 m^2/ha) of trees >10 cm diameter at breast height was lower in thinned than in unthinned plots. Data were collected in 2003-2005 in ten 0.1 ha plots in each of three replicates of thinned (low thinning and improvement/selection cutting) and unthinned 9 ha treatment units. Thinning was conducted in winter 2001.

A replicated, randomized, controlled study in 1976-2008 in boreal forest in Maine, USA (16) found that **thinning decreased tree density and basal area**. Number of trees/acre was higher in unthinned (2,962) than row-thinned (2,279) and the lowest in tree-released (1,699) and combined treatments (1,716). Basal area (ft^2/acre) was higher in unthinned (229) than tree-release (188) and combined treatments (173), and higher in row-thinned (206) than combined treatments (173). Data were collected in 2008 in four replicates of unthinned, row-thinned (5 ft. wide row removal with 3 ft. wide residual strips), tree-release (cutting selected trees at 8 ft. intervals) and combined (row-thinned plus tree-release) treatment plots (64 × 64 ft.) established in 1976.

A replicated, controlled study in 1997-2010 in Douglas-fir *Pseudotsuga menziesii* forest in Oregon, USA (17) found that **thinning decreased the density of mature trees**. Density of trees >5 cm diameter at breast height (unthinned: 531; thinned: 261-329 individuals/ha) and their basal area (unthinned: 61;

thinned: 35-45 m^2/ha) were higher in unthinned than in thinned plots. One unthinned and three thinned (retaining 100-300 trees/ha) treatment units (14-58 ha) were replicated in seven sites. Mature trees were monitored in 14-21 plots (0.1 ha) in each treatment unit. Treatments were applied in 1997-1999. Monitoring was 11 years after treatments.

A replicated, controlled study in 2001-2003 in temperate coniferous forest in California, USA (18) found **no effect of thinning and mulching on tree mortality casued by bark beetle**. Mortality of trees 11-25 cm DBH (thinned: <0.1%; unthinned: 0-0.2%), trees 25-45 cm DBH (thinned: 0%; unthinned: <0.1%) and trees >45 cm DBH (thinned: <0.1%; unthinned: <0.1%) was similar between treatments. Mortality caused by bark beetle for white fir *Abies concolor*, sugar pine *Pinus lambertiana* and ponderosa pine *Pinus ponderosa* trees was monitored in 2003 in 20 subplots (0.4 ha) in each of three unthinned and three thinned (crown thinning followed by thinning-from-below and mulching in 2001) treatment plots (14-29 ha).

(1) Schacht, W.H., Long, J.N., and Malechek, J.C. (1988) Above-ground production in cleared and thinned stands of semiarid tropical woodland, Brazil. *Forest Ecology and Management,* 23, 201-214.

(2) Sullivan, T.P., Sullivan, D.S., Lindgren, P.M.F., and Boateng, J.O. (2002) Influence of conventional and chemical thinning on stand structure and diversity of plant and mammal communities in young lodgepole pine forest. *Forest Ecology and Management,* 170, 173-187.

(3) Sendak, P.E., Brissette, J.C., and Frank, R.M. (2003) Silviculture affects composition, growth, and yield in mixed northern conifers: 40-Year results from the Penobscot Experimental Forest. *Canadian Journal of Forest Research,* 33, 2116-2128.

(4) Connell, M.J., Raison, R.J., and Jenkins, P. (2004) Effects of thinning and coppice control on stand productivity and structure in a silvertop ash (*Eucalyptus sieberi* L.Johnson) forest. *Australian Forestry,* 68, 30-38.

(5) Stephens, S.L., and Moghaddas, J.J. (2005) Experimental fuel treatment impacts on forest structure, potential fire behavior, and predicted tree mortality in a California mixed conifer forest. *Forest Ecology and Management,* 215, 21-36.

(6) Perchemlides, K.A., Muir, P.S., and Hosten, P.E. (2008) Responses of chaparral and oak woodland plant communities to fuel-reduction thinning in southwestern Oregon. *Rangeland Ecology and Management,* 61, 98-109.

(7) Brockway, D.G., Outcalt, K.W., Estes, B.L., and Rummer, R.B. (2009) Vegetation response to midstorey mulching and prescribed burning for wildfire hazard reduction and longleaf pine (*pinus palustris* mill.) ecosystem restoration. *Forestry,* 82, 299-314.

(8) Götmark, F. (2009) Experiments for alternative management of forest reserves: Effects of partial cutting on stem growth and mortality of large oaks. *Canadian Journal of Forest Research,* 39, 1322-1330.

(9) Harrod, R.J., Peterson, D.W., Povak, N.A., and Dodson, E.K. (2009) Thinning and prescribed fire effects on overstory tree and snag structure in dry coniferous forests of the interior Pacific Northwest. *Forest Ecology and Management,* 258, 712-721.

(10) Vaillant, N.M., Fites-Kaufman, J., Reiner, A.L., Noonan-Wright, E.K., and Dailey, S.N. (2009) Effect of fuel treatments on fuels and potential fire behavior in California, USA, national forests. *Fire Ecology*, 5, 14-29.

(11) Battaglia, M.A., Rocca, M.E., Rhoades, C.C., and Ryan, M.G. (2010) Surface fuel loadings within mulching treatments in Colorado coniferous forests. *Forest Ecology and Management,* 260, 1557-1566.

(12) Cole, E.C., Hanley, T.A., and Newton, M. (2013) Influence of precommercial thinning on understory vegetation of young-growth Sitka spruce forests in southeastern Alaska. *Canadian Journal of Forest Research,* 40, 619-628.

(13) Egan, J.M., Jacobi, W.R. Negron, J.F., Smith, S.L. and Cluck, D.R. (2010) Interacting effects of canopy gap, Forest thinning and subsequent bark beetle-caused mortality in Northeastern California. *Forest Ecology and Management,* 260, 1832-1842.

(14) Fettig, C., Borys, R., and Dabney, C. (2010) Effects of fire and fire surrogate treatments on bark beetle-caused tree mortality in the Southern Cascades, California. *Forest Science,* 56, 60-73.
(15) Fiedler, C.E., Metlen, K.L., and Dodson, E.K. (2010) Restoration treatment effects on stand structure, tree growth, and fire hazard in a ponderosa pine/douglas-fir forest in Montana. *Forest Science,* 56, 18-31.
(16) Weiskittel, A.R., Kenefic, L.S., Li, R., and Brissette, J. (2011) Stand structure and composition 32 years after precommercial thinning treatments in a mixed northern conifer stand in central Maine. *Northern Journal of Applied Forestry,* 28, 92-96.
(17) Dodson, E.K., Ares, A., and Puettmann, K.J. (2012) Early responses to thinning treatments designed to accelerate late successional forest structure in young coniferous stands of Western Oregon, USA. *Canadian Journal of Forest Research,* 42, 345-355.
(18) Stark, D.T., Wood, D.L., Storer, A.J., and Stephens, S.L. (2013) Prescribed fire and mechanical thinning effects on bark beetle caused tree mortality in a mid-elevation Sierran mixed-conifer forest. *Forest Ecology and Management*, 306, 61-67.

5.1.2. Thin trees within forests: effects on young trees

- Six of 12 studies (including two replicated, randomized, controlled studies) in Japan[2,4] and the USA[1,5-13] found that thinning trees in forests increased the **density of young trees**[1,2,5,9,10,11]. One study found that thinning decreased the density of young trees[11]. Five found no effect[4,7,8,13] or mixed effects[6] on the density of young trees. One replicated, controlled study in the USA[9] found no effect of thinning on the density of oak acorns.
- One controlled study in Peru[3] found that thinning increased the **growth rate of young trees.**

A replicated, controlled, paired site study in 1993–1995 in temperate coniferous Douglas-fir *Pseudotsuga menziesii* forest in western Oregon, USA (1) found that **thinning increased conifer seedling density**. Tree seedling density in thinned forest segments (1,433/ha) was greater than in unthinned forest segments (233/ha). Monitoring was in 1993–1995 in 32 pairs of thinned (between 1969 and 1984) and unthinned sites that had regenerated naturally following harvest between 1880 and 1940. Undisturbed old-growth Douglas-fir stands (>200 years) were present for comparison on 20 of the 32 paired sites.

A replicated controlled study in 1996-1997 in Japanese beech *Fagus crenata* forest in Japan (2) found that thinning **increased the number of new tree stems**. The number of new stems/ha (thinned: 686; unthinned: 413) was higher in thinned than unthinned plots. Data were collected in 1997 in 60 quadrats (5 × 5 m) in each of 17 thinned (30–70% by volume of the trees cut 10 years before measurements) and five unthinned plots (10 × 150 m).

A controlled study in 1989-2000 in tropical rainforest in Peru (3) found that five years after strip-clearing, **thinning enhanced annual growth increase of new tree stems.** In one cleared strip, annual growth increase for stems of three groups: recruits, stump sprouts and commercial species advance regeneration, was higher after thinning (0.13-0.19 cm) than in control plots (0.04-0.08 cm). In the other strip, annual growth increase for stems of commercial recruits, commercial stump sprouts, other recruits and other stump sprouts, was higher after thinning (0.20-0.28 cm) than in control plots (0.09-0.16 cm). Two 30×150 m strips were clear-cut in 1989. Each strip was divided into twenty 15×15 m plots. In 1996 all trees were thinned in two 30×45 m blocks in each strip. Data were collected in 2000.

A controlled study in 1997-2001 in temperate coniferous forest in Japan (4) found **no effect of thinning on Japanese black pine** *Pinus thunbergii* **seedlings density**. Density (seedlings/m^2) was similar in all thinning treatments (unthinned: 14; 20%: 16, 30%: 13; 50%: 17). Four treatments: unthinned and 20%, 30% and 50% of the area thinned in a patch pattern were applied each to a 40 × 50 m forest section in December 1997. Japanese black pine seedlings were monitored four growing seasons after thinning in five 2 × 2 m plots in each treatment.

A replicated, controlled study in 1999-2003 in temperate mixed forest in California, USA (5) found that **thinning by removal of all conifers increased trembling aspen** *Populus tremuloides* **density**. Total aspen density (stems/ha) was higher in thinned (16,000) than in unthinned plots (6,000). Data were collected in 2003 in 2-4 transects (30.5 × 1.8 m) in each site (~1.7 ha) of four thinned (all conifers removed in 1999) and unthinned pairs.

A replicated, controlled study in 2001-2005 in second-growth oak *Quercus spp*. forests in southern Ohio, USA (6) found that **mechanical thinning reduced small seedling density and increased large seedling and small sapling densities.** Density (individuals/ha) of small (<50 cm tall) seedlings was lower in thinned plots (unthinned: 135,000; thinned: 70,000). In contrast, the density of large seedlings (40-150 cm tall) (unthinned: 2,000; thinned: 7,000) and small saplings (<3 cm DBH) (unthinned: 1,000; thinned: 2,400) was higher in thinned plots. Thinning had no effect on density of large saplings (3-10 cm DBH) (unthinned: 600; thinned: 500). Three forest areas were divided into unthinned and thinned (mechanical-thinning) treatment units (30 ha). Treatments were applied in the inactive season of 2001. Regeneration was sampled in ten 0.1 ha plots in each treatment (a total of 40 plots/site) in summer 2004.

A replicated, controlled study in 2000-2005 in temperate broadleaf forest in Ohio, USA (7) found **no effect of thinning on numbers of black oak** *Quercus velutina* **and chestnut oak** *Q. prinus* **acorns**. The density (acorns/ha) of black oak (20,000-30,000) and chestnut oak (30,000-40,000) was similar between treatments. Data were collected in 2005 in nine thinned (thinning from below retaining 70% of tree basal area in 2000-2001) and nine unthinned plots (0.1 ha) at each of two sites.

A replicated, randomized, controlled study in 2000-2005 in temperate forest in California, USA (8) found **no effect of thinning followed by mulching on conifer or California black oak *Quercus kelloggii* seedling densities**. The combined density (trees/m^2) of conifer and oak seedlings (thinned: 2.8; unthinned: 1.5) and the density of oaks (thinned: 0.40; unthinned: 0.45) were similar between treatments. Data were collected in 2006 in 10 sets of four plots (1 m^2) in each of three thinned (trees >25 cm DBH removed in 2001, followed by mulching of the remaining trees) and three unthinned treatment units (14-29 ha).

A replicated, controlled, before-and-after study in 2000-2004 in Piedmont forest in South Carolina, USA (9) found that **thinning increased tree seedling density**. Changes in density of tree seedlings <1.4 m tall was higher in thinned plots (thinned: 19,400/ha; unthinned: 8,550/ha). Changes in density of tree saplings >1.4 m tall and <10 cm DBH were similar between treatments (thinned:

515; control: 243). Ten plots (0.1 ha) were established in 2000/2001 in each of three unthinned and three thinned (basal area reduced to 18 m^2/ha) treatment units. Data were collected three years after treatment.

A replicated, randomized, controlled study in 2000-2007 in temperate broadleaf forest in North Carolina and Ohio, USA (10) found that **thinning trees increased the cover of seedlings and the density of tree saplings**. At a 'cool temperate climate' site the number of hardwood-tree saplings (>1.4 m tall) (thinned: 800/ha, unthinned: 370/ha) and cover of shrub and tree seedlings (< 1.4 m tall) (thinned: 53%, unthinned: 27%) were higher in thinned than unthinned plots. At a 'warm continental climate' site, cover of shrub and tree seedlings was higher in thinned plots (thinned: 28%, unthinned: 18%), while numbers of tree saplings was similar between treatments (thinned: 1200/ha, unthinned: 1800/ha). Three pairs of thinned (in 2000-2002) and unthinned treatment units (10-26 ha) were established at each of the two sites. Data were collected 4-5 years post-treatments in 10 plots (0.1 ha) in each treatment unit.

A replicated, controlled study in 2001-2005 in temperate coniferous forest in Montana, USA (11) found that **thinning decreased tree sapling density**. The density of tree-saplings >0.1 and <10 cm diameter at breast height was lower in thinned (5,293 stems/ha) than in unthinned plots (11,483 stems/ha). Data were collected in 2003-2005 in ten 0.1 ha plots in each of three replicates of thinned (low thinning and improvement/selection cutting) and unthinned 9 ha treatment units. Thinning was conducted in winter 2001.

A replicated, controlled study in 1997-2010 in Douglas-fir *Pseudotsuga menziesii* forest in Oregon, USA (12) found that **thinning increased the density of new tree stems**. The density of saplings >137 cm tall and <5 cm diameter at breast height (unthinned: 114; thinned: 527-815) and seedlings 15-136 cm tall (unthinned: 502; thinned: 2,719-4,594) was higher in thinned than in unthinned plots. One unthinned and three thinned (retaining 100-300 trees/ha) treatment units (14-58 ha) were replicated in seven sites. Saplings and seedlings were monitored in four subplots (0.002 ha) within each plot. Treatments were applied in 1997-1999. Monitoring was 11 years after treatments.

A before-and-after study in 2003-2005 in temperate coniferous forest in California, USA (13) found **no effect of thinning on the density of conifer seedlings and saplings**. There was no difference between treatments for the change in density (individuals/ha after minus before) of seedlings <1.37 m tall (thinned: -539; unthinned: -2,303) and saplings >1.37 m tall and <10 cm DBH (thinned: -222; unthinned l: 74). Data were collected in 2003 (before) and 2005 (after) in five plots (0.04 ha) in each of two thinned (thinned to retain 30 m^2/ha basal area with slash mulching in June 2003) and two unthinned treatment units (~1 ha).

(1) Bailey, J.D., and Tappeiner, J.C. (1998) Effects of thinning on structural development in 40- to 100-year, old Douglas-fir stands in western Oregon. *Forest Ecology and Management,* 258, 398-408.

(2) Nagaike, T., Kamitani, T., and Nakashizuka, T. (1999) The effect of shelterwood logging on the diversity of plant species in a beech (Fagus crenata) forest in Japan. *Forest Ecology and Management,* 118, 161-171.

(3) Dolanc, C.R., Gorchov, D.L., and Cornejo, F. (2003) The effects of silvicultural thinning on trees regenerating in strip clear-cuts in the Peruvian Amazon. *Forest Ecology and Management,* 182, 103-116.

(4) Zhu, J., Matsuzaki, T., Lee, F., and Gonda, Y. (2003) Effect of gap size created by thinning on seedling emergency, survival and establishment in a coastal pine forest. *Forest Ecology and Management,* 182, 339-354.
(5) Jones, B.E., Rickman, T.H., Vasquez, A., Sado, Y., and Tate, K.W. (2005) Removal of encroaching conifers to regenerate degraded aspen stands in the Sierra Nevada. *Restoration Ecology,* 134, 373-379.
(6) Albrecht, M.A., and McCarthy, B.C. (2006) Effects of prescribed fire and thinning on tree recruitment patterns in central hardwood forests. *Forest Ecology and Management,* 226, 88-103.
(7) Lombardo, J.A., and McCarthy, B.C. (2008) Silvicultural treatment effects on oak seed production and predation by acorn weevils in southeastern Ohio. *Forest Ecology and Management,* 225, 2566-2576.
(8) Moghaddas, J.J., York, R.A., and Stephens, S.L. (2008) Initial response of conifer and California black oak seedlings following fuel reduction activities in a Sierra Nevada mixed conifer forest. *Forest Ecology and Management*, 255, 3141-3150.
(9) Phillips, R.J., and Waldrop, T.A. (2008) Changes in vegetation structure and composition in response to fuel reduction treatments in the South Carolina Piedmont. *Forest Ecology and Management,* 255, 3107-3116.
(10) Waldrop, T.A., Yaussy, D.A., Phillips, R.J., Hutchinson, T. A., Brudnak, L., and Boerner, R.E.J. (2008) Fuel reduction treatments affect stand structure of hardwood forests in Western North Carolina and Southern Ohio, USA. *Forest Ecology and Management,* 225, 3117-3129.
(11) Fiedler, C.E., Metlen, K.L., and Dodson, E.K. (2010) Restoration treatment effects on stand structure, tree growth, and fire hazard in a ponderosa pine/douglas-fir forest in Montana. *Forest Science,* 56, 18-31.
(12) Dodson, E.K., Ares, A., and Puettmann, K.J. (2012) Early responses to thinning treatments designed to accelerate late successional forest structure in young coniferous stands of Western Oregon, USA. *Canadian Journal of Forest Research,* 42, 345-355.
(13) Walker, R.F., Fecko, R.M., Frederick, W.B., Johnson, D.W., and Miller, W.W. (2012) Seedling recruitment and sapling retention following thinning, chipping, and prescribed fire in mixed Sierra Nevada conifer. *Journal of Sustainable Forestry,* 31, 747-776.

5.1.3. Thin trees within forests: effects on understory plants

- Seventeen of 25 studies (including four replicated, randomized, controlled studies) in Argentina[32], Brazil[1], Canada[2,7,11], Japan[4], Spain[3] and the USA[5,10,13,16,19,20,22,23,24,25,26,29,30,31,34,35,36,37] found that thinning trees in forests increased the **density and cover of understory plants**[1,4,5,10,11,13,19,20,22,24,26,29,30,32,34,35,37]. Seven studies found no effect[7,16,23,25,36] or mixed effects[3,31]. One study[2] found a decrease in the abundance of herbaceous species.
- Thirteen of 19 studies (including 10 replicated, randomized, controlled studies) in Argentina[32], Canada[7,17], Sweden[8], the USA[5,6,9,12,14,15,16,18,20,22,27,28,29,33,37] and West Africa[21] found that thinning trees in forests increased **species richness and diversity of understory** plants[5,6,8,9,14,17,20,22,27,28,29,32,33]. Seven studies found no effect[7,12,15,16,18,21,37].

A replicated, controlled study in 1984-1985 in dry tropical forest in Ceara state, Brazil (1) found that **thinning trees increased herbaceous plant biomass.** Biomass of herbaceous species that matured late in the season was the lowest in unthinned plots (0% tree cover: 1,649; 25% cover: 1,593; 55% cover: 1,600; unthinned: 221), while total herbaceous biomass was similar between treatments (0% cover: 1,981; 25% cover: 1,845; 55% cover: 1,926; unthinned: 259). Four treatment plots (0.1 ha) were established in 1984 in each of two sites: three thinned (0%, 25% and 55% woody cover retained) and one unthinned

(95% woody cover). Data were collected in May 1985 in a subplot protected from grazing (40 × 50 m) in each plot.

A replicated, controlled study in 1985-1988 in boreal forest in Ontario, Canada (2) found that **thinning decreased the number of herbaceous species and the frequency of occurrence of each species**. In large plots (0.05 and 0.2 ha), the percentage of herbaceous species that decreased in frequency was higher in uncut (34%-36%) than in 33% tree removal (16-18%) and 66% tree removal plots (18-21%). In contrast, in small plots (0.1 ha), figures were higher in uncut (37%) and 33% tree removal plots (36%) than 66% tree removal plots (12%). The percentage of herbaceous species lost was similar between treatments (uncut: 9-13%; 33% removal: 4-12%; 66% removal: 8-12%). Three plots (0.01, 0.05 and 0.20 ha) of each treatment were replicated five times: uncut, 33% tree removal and 66% tree removal (0%, 33%, and 66% of basal area removed). Treatments were applied in 1985-1986. Data were collected two years after treatments.

A replicated, controlled study in 1991-1994 in maritime pine *Pinus pinaster* woodland in Spain (3) found that **thinning before wildfire increased post-fire biomass and species richness of herbaceous species, but not of the dominant shrub gum rockrose *Cistus ladanifer***. Herbaceous biomass (g/m^2) (pre-thinned: 37-93; unthinned: 2-10) and species richness (species/plot) (pre-thinned: 6-16; unthinned: 5-7) were higher in pre-thinned plots. Herbaceous cover (pre-thinned: 13%-49%; unthinned: 3%-11%) and **gum rockrose** cover (pre-thinned: 8%-46%; unthinned: 16%-32%) and density () (pre-thinned: 1-10/m^2; unthinned: 2-7/m^2) were similar between treatments. Data were collected in six thinned (1975-1991) and six unthinned plots (5 × 10 m), three years after the entire study site was burned by wildfire fire in 1991.

A replicated, controlled study in 1996-1997 in Japanese beech *Fagus crenata* forest in Japan (4) found that **thinning increased the occurrence of dwarf bamboo *Sasa sp.*** The percentage occurrence of dwarf bamboo was higher in thinned plots (thinned: 59%; unthinned: 44%). Data were collected in 1997 in 60 quadrats (5 × 5 m) in each of 17 thinned (30–70% by volume of the trees cut 10 years before measurements) and five unthinned plots (10 × 150 m).

A replicated, controlled study in 1999 in temperate mixed forest in Arizona USA (5) found that **thinning increased the abundance of native grasses but did not affect species richness for any under-canopy plant group**. Abundance index of native grass species was higher in thinned (33) than in unthinned plots (19). Abundance index of native herbaceous species (23 vs 26), exotic herbaceous species (1 vs 3) and exotic grasses (4 vs 0), and the number of species (/375 m^2) of native herbaceous species (17 vs 18), exotic herbaceous species (2 in both), native grasses (6 in both) and exotic grasses (1 vs 0) were similar between thinned and unthinned plots. Data were collected in ten 375 m^2 plots in each of four thinned (30% of basal area removed between 1987 and 1993) and four unthinned forest fragments (20-80 ha).

A replicated, randomized, controlled study in 1993-1996 in temperate coniferous forest in Washington State, USA (6) found that **variable density thinning increased plant species richness and diversity and the proportion of exotic plant species.** Total species richness (thinned: 24-27; unthinned: 16-17 species/100 m^2 plot), native species richness (thinned: 21-22; unthinned: 15-17), Shannon's index of diversity (thinned: 2.5-2.7; unthinned: 1.9-2.0) and the percent of exotic species (thinned: 12%-17%; unthinned: 2%) were higher in thinned plots. Two thinned (variable density thinning to a 2:1 ratio of >4.75 and <4.75 residual trees/ha respectively) and two unthinned treatment units (13 ha) were established in 1993 in each of four sites. Data were collected in 1994 and 1996 in 15 plots (25 m^2) in each treatment unit.

A replicated, controlled study in 1993-1998 in temperate lodgepole pine *Pinus contorta* forest in British Columbia, Canada (7) found **no effect of lodgepole pine thinning on total plant species richness**. The number of plant species/treatment unit was similar between treatments (thinned: 22; unthinned: 23). Data were collected in 1998 in thinned (targeted to retain 1,000 stems/ha) and unthinned treatment units (1.8-12.6 ha) established in 1993 in each of three study areas.

A replicated, controlled, before-and-after study in 2000-2003 in temperate broadleaf forest in Sweden (8) found that **thinning trees increased species richness of herbaceous species**. The increase in herbaceous species richness was higher in thinned (18.3%) than in unthinned plots (1.2%). Average numbers of species/25 m^2 section was 13-27 before vs 14-29 after treatment in thinned plots, and 13-28 before vs 13-26 after treatment in unthinned plots. Thinned (25-30% of basal area cut) and unthinned treatments were applied to six pairs of 1 ha plots in winter 2002-2003. Data were collected before (2001-2002) and after treatment (summer 2003) in eight sections (25 × 1 m) within each plot.

A replicated, randomized, controlled study in 2001-2004 in temperate coniferous forest in Montana, USA (9) found that **thinning increased understory plant species richness**. Numbers of species/0.1 ha plot for all species (unthinned: 57; thinned: 66) as well as for native species (unthinned: 53; thinned: 59), exotic species (unthinned: 4; thinned: 7) and forbs (unthinned: 34; thinned: 40) was higher in thinned plots. Numbers of species of grasses graminoids (12-14) and shrubs (9-10) were similar between treatments. Numbers of species/1 m^2 was higher in thinned plots for forbs (unthinned: 5.5; thinned: 6.4) and similar between treatments for all species (10.8- 12.2) and for the other plant groups (native species: 10.5-11.8; exotic species: 0.3-0.4; graminoids: 2.4; shrubs: 2.4-2.9). Cover of all plants (28-32%) was similar between treatments. In 2001, ten plots (0.1 ha) were established in each of three replicates of thinned (retaining 11 m^2/ha basal area) and unthinned treatment units (9 ha). Species composition was determined in 2004 in 12 quadrats (1 m^2) in each plot (total of 720 quadrats).

A replicated, controlled study in 1992-2004 in Ponderosa pine *Pinus ponderosa* forest in Arizona, USA (10) found that **thinning increased herbaceous biomass**. Herbaceous biomass (kg/ha) was higher in thinned (270-280) than in unthinned plots (~10). Data were collected in 2004 in four circular subplots (2.5 m radius) in each of 10 thinned (thinned from below in 1993, retaining trees 40.6 cm DBH) treatment plots (0.2-0.3 ha), and in three subplots in each of five unthinned treatment plots (total of 55 subplots).

A replicated, controlled study in 2000-2002 in boreal forest in Alberta, Canada, (11) found that **removal of trembling aspen *Populus tremuloides* canopies increased the biomass of understory vegetation and cover of herbaceous species**. Biomass (kg/ha) of understory vegetation was higher in partial (1,300-2,200) and complete removal plots (2,100-2,700) than control plots (700-850) at the parkland site and differed between all treatments at the boreal site (control: 400-750; partial removal: 1,100-1,150; complete removal: 2,100-2,900). Cover of non-grass herbaceous plants at the boreal site (control: 29-45%; partial removal: 33-38%; complete removal: 46-66%) and of grasses at the parkland site (control: 8-20%; partial removal: 15-37%; complete removal: 52-79%) was higher in complete than in partial removal and control plots. Cover of tall shrubs (>1 m) at the boreal site was lower in partial (3-8%) and complete removal (8-20%) than in control plots (15-42%). There was no difference between treatments for the following: cover of tall shrubs at the parkland site (control: 4-10%; partial removal: 5-8%; complete removal: 0-3%), low shrubs (<1 m) at the parkland (control: 25-31%; partial removal: 12-25%; complete removal: 17-38%) and at the boreal site (control: 24-51%; partial removal: 35-40%; complete removal: 46-52%), forbs at the parkland (control: 7-10%; partial removal: 3-4%; complete removal: 4-7%) and grasses at the boreal site (control: 1-3%; partial removal: 0-2%; complete removal: 2-9%). Three replicates of complete removal (all aspen canopies removed), partial removal (half of aspen canopy area removed) and control plots (10 × 10 m) were established in 2000 in a 'boreal' site (16,319 stems/ha) and a 'parkland' site (13,194 stems/ha). Data were collected in 2002.

A replicated, randomized, controlled study in 1998-2004 in temperate coniferous forest in Oregon, USA (12) found **no effect of thinning on understory species richness and diversity**. Numbers of species/400 m^2 plot (thinned: 26; unthinned: 30) and diversity (Shannon's index thinned: 0.12; unthinned: 0.12) were similar between treatments. Data were collected in 2004 in 10-28 plots (400 m^2) in each of four thinned (thinned in 1998 to reduce trees basal area from 26 to 16 m^2/ha) and four unthinned experimental units.

A replicated, controlled study in 2001-2005 in temperate mixed forest in Washington State, USA (13) found that **conifer cutting increased cover of non-native, but not of native plants under Oregon white oak *Quercus garryana* canopies**. Under oak canopies cover of non-native forbs (conifer cut: 10%; uncut: 7%), grasses (conifer cut: 24%; uncut: 12%) and woody plants (conifer cut: 20%; uncut: 9%) was higher under conifer cut oaks. There was no difference between treatments for cover of native forbs (conifer cut: 30%; uncut: 35%), grasses (conifer cut: 7%; uncut: 5%) and woody plants (conifer cut: 127%; uncut: 128%), or total plant cover under Oregon white oak canopies (99% under both conifer cut and uncut oak trees). Data were collected in 2005 under six conifer cut (all conifer covering the oaks cut in 2001) and six control Oregon white oak trees (average height: 16 m, average crown diameter 7.5 m) at each of four forest sites.

A replicated, randomized, controlled study in 2001-2004 in temperate conifer forest in Montana, USA (14) found that **thinning increased native plant species richness**. Species richness (in 1,000 m^2) for common (thinned: 34; unthinned: 32) and uncommon (thinned: 15; unthinned: 12) native plant species was higher in thinned plots. Data were collected in 2004 in 10 thinned (in 2001, 11 m^2/ha retained) and 10 unthinned plots (1000 m^2) in each of three blocks.

A replicated randomized, controlled study in 2000-2004 in temperate broadleaf forest in Ohio, USA (15) found **no effect of thinning on soil seed-bank species richness or diversity**. Total numbers of species (thinned: 37; unthinned: 38) and Shannon's index of diversity (thinned: 3.03; unthinned: 3.11) were similar between treatments. In autumn to winter 2000-2001, ten plots (20

× 50 m) were established within each thinned (retaining ~13.75 m^2/ha basal area) and unthinned treatments (20 ha) replicated at each of two sites. Species richness and diversity were determined by monitoring emerging seeds in 10 soil samples (1000 cm^3) extracted from each plot in summer 2004.

A replicated, randomized, controlled study in 2000-2003 in temperate mixed forest in California, USA (16) found **no effect of thinning on understory plant species richness and cover**. Numbers of species/10 m^2 plot (unthinned: 4; understory thinning: 4; canopy thinning: 3) and cover (unthinned: 8%; understory thinning: 5%; canopy thinning: 6%) were similar among treatments. Three replicates of unthinned, understory thinning (removing trees 25–76 cm DBH, retaining at least 40% canopy cover) and canopy thinning (removing trees >25 cm DBH leaving 22 large trees/ha) treatment units (4 ha) were established in 2000-2001. Data were collected in 2003 in 9-49 plots (10 m^2) in each treatment unit.

A replicated, controlled, before-and-after trial in 2004-2005 in temperate broadleaf forest in Ontario Canada (17) found that **thinning increased the species richness of herbs**. The increase in number of herbaceous species/plot was higher in thinned (3.6 to 4.5) than in unthinned plots (4.3 to 4.8). Overall percent of plant species lost (15% and 11% unthinned and thinned respectively) and of plant species gained (29% and 42%) was similar among treatments. Two thinned (leaving basal area of 20 m^2/ha) and two unthinned blocks (average 33 ha) were established between November 2004 and April 2005. Sampling of herbs that grew mid-spring was in April 2004 (pre-harvesting) and in April-May 2005 (post-harvesting) in 45 regeneration growth plots (4 m^2) in each block.

A replicated, controlled study in 1992-2005 in temperate coniferous forest in Arizona, USA (18) found **no effect of thinning on plant species richness or on changes in species composition**. Numbers of species/2 m^2 (unthinned: 6; thinned: 8) and the change in species composition between 1992 and 2005 (unthinned: 0.36; thinned: 0.44) were similar between treatments. Complete species lists were collected in two 1 m^2 quadrats in each of 35 subplots (2.5 m^2), four in each of five thinned (thinned from below in 1993, retaining all trees >37.5 cm DBH) and three in each of five unthinned plots (0.2-0.3 ha). Data were collected between 1992 and 2005.

A replicated, paired sites study in 2005 in Mediterranean type woodland in Oregon, USA (19) found that **thinning trees increased the cover of herbs and the number of regenerating shrubs**. Cover of herbs (thinned: 103%; unthinned: 69%) and number of the shrubs sticky whiteleaf manzanita *Arctostaphylos viscida* and buckbrush *Ceanothus cuneatus* regenerations/transect (thinned: 1.7; unthinned: 0.3) were higher in thinned transects. Plant species richness/transect (thinned: 29; unthinned: 28) and diversity (Shannon's index thinned: 2.3; unthinned: 2.4), as well as number of regenerations/transect of oak *Quercus spp.* (thinned: 1.7; unthinned: 2.1) and conifer (thinned: <0.1; unthinned: <0.1) were similar in thinned and unthinned transects. Data were collected in 2005 using 30 pairs of thinned (thinned for fuel reduction between May 1998 and June 2001) and unthinned transects (50 m). Shrub cover was measured in five plots (3 m^2) along each transect. Cover of herbs was measured in two quadrats (1000 cm^2) within each plot.

A replicated, controlled, before-and-after study in 2000-2004 in Piedmont forest in South Carolina, USA (20) found that **thinning increased plant species**

richness. Changes (after minus before treatment) in number of plant species/0.1 ha plot were higher in thinned plots (thinned: 39; unthinned: 32). Changes in cover of shrubs (thinned: 0.27%; unthinned: -0.41%), vines (thinned: 0.09%; unthinned: -2.73%), forbs (thinned: 0.29%; unthinned: 0.22%) and grasses (thinned: 0.52%; control: -0.48%) were similar between treatments. Ten plots (0.1 ha) were established in 2000-2001 in each of three unthinned and three thinned (basal area reduced to 18 m^2/ha) treatment units. Data were collected three years after treatment.

A replicated, randomized, controlled study in 1994-2003 in savanna woodland in West Africa (21) found **no effect of cutting on species richness or diversity of herbs**. Numbers of species/0.25 ha (uncut: 13-16; cut: 14-16) and diversity (Shannon's index uncut: 2.5-2.9; cut: 2.6-2.9) was similar between treatments. Data were collected in 2003 in two uncut and two cut (50% of merchantable tree volume removed in 1994) treatment plots (0.25 ha) replicated in eight blocks, at each of two sites (18 ha).

A replicated, controlled study in 2000-2006 in temperate mixed forest in Vermont and New York, USA (22) found that **thinning increased species richness, diversity and cover of understory plants**. Changes in number of species/0.04 plot (control: -1; group: 0; single tree cutting: 4; complexity enhancement cut: 9) and cover (control: -5%; group cut: 2%; single tree cutting: -4%; complexity enhancement cut: 8%) were higher in complexity enhancement than control plots. Change in diversity was higher in complexity enhancement cuts (Shannon's index: 3) and single tree cutting (2) than control plots (-0.5). Eight control (unthinned), four single tree cutting (cutting in dispersed pattern, retaining 18.4 m^2/ha basal area), four group cut (cutting in aggregated pattern, retaining 18.4 m^2/ha basal area) and four complexity enhancement cuts (cutting trees to a target typical diameter distribution, retaining 34 m^2/ha basal area) treatment units (2 ha) were established in 1999-2003. Data were collected three years after treatments in eight plots (0.04 ha) in each treatment unit.

A replicated, randomized, controlled study in 2000-2007 in temperate broadleaf forest in North Carolina and Ohio, USA (23) found **no effect of tree thinning on herbaceous cover.** At a 'cool temperate climate' site the number of hardwood tree saplings (>1.4 m tall) and cover of herbs (thinned: 3-19%, unthinned: 5-13%) were similar between treatments. Three pairs of thinned (in 2000-2002) and unthinned treatment units (10-26 ha) were established at each of two sites. Data were collected 4-5 years post-treatments in 10 plots (0.1 ha) in each treatment unit.

A replicated, controlled study in 2000-2003 in temperate mixed forest in Georgia, USA (24) found that **mechanical thinning increased the cover of understory plants**. Understory plant cover was higher in thinned than unthinned plots (thinned: 112%; unthinned: 71%). Four blocks, each containing thinned (mulching of all broadleaf trees regardless of size, and all pines <20 cm diameter at breast height) and unthinned treatment plots (110 × 110 m) were established in 2000. Data were collected in 2002-2003 in five subplots (10 × 10 m) within each treatment plot.

A replicated, controlled study in 1988-2005 in temperate coniferous forest in Arizona, USA (25) found **no effect of thinning on understory plant biomass**. Above ground biomass (kg/ha) of native grasses (unthinned: 600; thinned: 1,100) and forbs (unthinned: 300; thinned: 250) was similar between treatments. No non-native grasses or forbs were found in control or thinned plots. Data were collected in 2005 in 10 plots (20 × 50 m) in each of three unthinned and four thinned (>30% of basal area removed between 1988 and 1995) forest units (20-80 ha).

A replicated, controlled study in 1994-2005 in temperate coniferous forest in Colorado, USA (26) found that **thinning increased understory vegetation cover and the proportion of non-native species**. Understory vegetation cover (unthinned: 3.9%; thinned: 6.1%; thinned and chipped: 7.1%) was higher in thinned and chipped plots than unthinned plots. The proportion of non-native understory species was higher in the thinning treatments (18% in both) than the unthinned treatment (14%), while the total number of species/1,000 m^2 was similar between treatments (unthinned: 53; thinned: 47; thinned and chipped: 48). Data were collected in 2005-2006 in 31 plots (1,000m^2) established in 1994. Six plots were unthinned, 13 thinned (harvested matter removed from the site) and 12 were thinned and chipped (harvested matter chipped and distributed on the site).

A replicated, randomized, controlled study in 1997-2008 in temperate coniferous forest in western Oregon, USA (27) found that **thinning increased the number of understory species**. The number of species/80 m^2 was higher following fixed (high or moderate) density thinning (76 and 86 respectively) than following variable (high or moderate) density thinning (54 and 55 respectively) and unthinned (48). It was not different than the other five treatments following variable low density (60). A set of six thinning regimes, each comprising 20–44 ha, was applied in 1997 at each of three forest sites: unthinned; fixed high density treatment (300 trees/ha); fixed moderate density treatment (200 trees/ha); variable high density treatment (300 trees/ha); variable moderate density treatment (200 trees/ha); variable low density treatment (100 trees/ha). Between four and 20 permanent 0.1 ha plots were located randomly in each treatment (total of 77 plots/site). Four 20 m^2 sub-plots were installed in each plot. Monitoring was carried out in summer 2003 and 2008.

A replicated, controlled study in 2002-2005 in an oak *Quercus spp.* savanna in Iowa, USA (28) found that **cutting all non-oak trees increased species richness**. Species richness/1 m^2 (non-oaks cut: 18; uncut: 10) as well as species richness of grasses (non-oaks cut: 3; uncut: 1) and woody plants (non-oaks cut: 8; uncut: 4) were higher in non-oaks cuts than in uncut plots. Diversity (Simpson's index non-oaks cut: 8; uncut: 5) and forb species richness (non-oaks cut: 7; uncut: 4) were similar between treatments. The percentage of native species was higher in uncut plots (non-oaks cut: 94%; uncut: 99%). Data were collected in 2004-2005 in 11-21 plots (1 × 1 m) at each of four non-oaks cut (all non-oak trees >1.5 m tall removed in 2002-2003) and four uncut sites (1.5-3.3 ha).

A replicated, randomized, controlled study in 1998-2005 in boreal forest in south eastern Alaska, USA (29) found that **thinning trees increased the cover of understory vegetation**. The total cover of understory plants was similarly higher in all thinning treatments (62-72%) than in unthinned plots (30%). Two 0.2 ha plots of each of four conifer thinning treatments (retaining 250, 370, 500, and 750 trees/ha) and unthinned plots were replicated in seven 16-18 year old forest sections. Treatments were applied in 1999, data were collected in 2005.

A replicated, controlled study in 2004-2008 in temperate broadleaf forest in Pennsylvania, USA (30) found that **tree thinning increased the cover of bramble *Rubus spp*. and fern as well as tree saplings density, but did not affect fruit production and cover of some herbaceous species**. Cover of bramble (thinned: 0%-27%; unthinned: 0%-3%) and hay-scented fern *Dennstaedtia punctilobula* (thinned: 0%-70%; unthinned: 0%-33%), as well as number of tree saplings/m^2 (thinned: 0.0-1.8; unthinned: 0.0-0.4) were higher in thinned plots. Total number of fruit/plot for three herbs: painted trillium *Trillium undulatum*, sessile bellworth *Uvularia sessilifolia*, and Indian cucumber root *Medeola virginiana* (0-430) as well as their relative cover (0-3%) were similar between treatments. Data were collected in 2008 in three blocks of 16 thinned (10-30% of basal area removed in 2001-2002) and eight unthinned plots (50 × 80 m) each.

A replicated, randomized, controlled study in 2002-2008 in temperate coniferous forest in Alabama, USA (31) found that **thinning decreased the density of understory shrubs and trees and increased the cover of grasses**. Density (stems/ha) of hardwood trees <3 cm DBH (thinned: <50; unthinned: >1,500) and cover of shrubs >1.4 m tall (thinned: <1%; unthinned: 33%) were higher in control plots, while cover of grasses (thinned: 20%; unthinned: 7%) was higher in thinned plots. Cover of shrubs <1.4 m tall (~55%) and forbs (3%-8%) were similar between treatments. Unthinned and thinned (leaving 11.5–13.5 m^2 basal area of longleaf pine *Pinus palustris*, removing hardwoods and other pines) treatment units (12 ha) were replicated in three blocks. Thinning was in April 2002. Data were collected in 2005 in ten 20 × 50 m subplots within each treatment unit.

A controlled study in 2001-2005 in temperate *Nothofagus pumilio* forest in Argentina (32) found that **thinning increased plant cover, biomass and species richness**. Cover (thinned: 36-40%; unthinned: 20%) and biomass (thinned: 1,000-1,251 kg/ha; unthinned: 200 kg/ha) of understory plants were higher in the three thinning treatments. Numbers of plant species/1 m^2 was higher in aggregated retention plots (8.2) than in unthinned plots (6.1), and similar to both in dispersed (7.1) and combined retention plots (7.0). In 2001, three thinning treatments (11-24 ha): dispersed retention (20–30% of green tree retention); aggregated retention (28% of trees retained, one aggregate of forest/ha); combined retention (40–50% of retention, one aggregate/ha and dispersed retention among them), and unthinned (9 ha) were established within a 61 ha area. Data were repeatedly collected 1-4 years after treatments in 10 permanent plots (1 m^2) in each treatment.

A replicated, randomized, controlled study in 1998-2006 in temperate coniferous forest in Arizona USA (33) found that **thinning increased plant species richness.** The number of observed species was higher in thinned (34-38) than unthinned plots (20), while plant cover was similar between treatments (thinned: 9-16%; unthinned: 4%). Monitoring was carried out in 2006 in three thinned and one unthinned 14 ha forest units that were randomly assigned in 1998 in each of three blocks.

A replicated study in 1975-2006 in temperate coniferous forest in Oregon USA (34) found that a **second thinning treatment increased the cover and abundance of some understory plant groups**. The percentage cover of ferns (43 vs 30%) and exotic plant species (1.0 vs 0.1%) was higher in twice thinned than in once thinned plots, while percentage cover of all forest understory species was similar between treatments (95% in both treatments). Frequencies were higher in twice thinned than in once thinned plots for ferns (2.0 vs 1.6 respectively), grasses (2.2 vs 1.3), open site species (4.1 vs 2.2) and exotic species (0.7 vs 0.1).The frequency of all forest understory species was similar between treatments (10.3 vs 9.9). Two treatments: once thinned (thinned from below in 1975-1982 to densities of 270-590 trees/ha) and twice thinned (re-thinned in 1997-2000 to 100-150 trees/ha) were replicated in four sites. Understory vegetation was monitored six years after the second thinning in 6-12 once thinned and 12-13 twice thinned 0.1 ha plots at each site.

A controlled study in 2007-2009 in Piñon-juniper woodland in Utah, USA (35) found that **thinning increased understory vegetation cover**. Cover of understory plants was higher in the two thinning treatments (piled and burned: 16%; woody debris: 21%) than control plots (4%). Three treatment sites (0.4-1 km^2): piled and burned (trees manually cut with debris placed in discrete piles that were later burned), woody debris (trees manually removed and debris scattered across the site) and control (untreated) were established in 2007. Data were collected in 2009 along 10 transects (35 m) in each site.

A before-and-after study in 2003-2005 in temperate coniferous forest in California, USA (36) found **no effect of thinning on understory vegetation cover**. The changes (after minus before) in cover (thinned: 0%; unthinned: -2%) were similar between treatments. Data were collected in 2003 (before) and 2005 (after) in five plots (0.04 ha) in each of two thinned (thinned to retain 30 m^2/ha basal area with debris mulched in June 2003) and two unthinned treatment units (~1 ha).

A replicated, controlled study in 2004-2011 in temperate coniferous forest in Arizona, USA (37) found that **thinning increased plant cover but not species richness**. Total plant cover was higher in thinned plots (thinned: 5.4%; unthinned: 3.1%), while species richness (33-37 species) and diversity (Simpson's index 0.8-0.9) were similar between treatments. Four thinned (pinyon pine *Pinus edulis* trees <25.4 cm diameter at root collar, Utah juniper *Juniperus osteosperma* <30 cm diameter at root collar and ponderosa pine *Pinus ponderosa* trees <22.9 cm diameter at breast height cut) and four unthinned treatment units (1 ha) were replicated in six blocks. Thinning was in 2005. Data were collected in 2011 in one 0.04 ha plot in each treatment unit (total of 48 plots).

(1) Schacht, W.H., Long, J.N., and Malechek, J.C. (1988) Above-ground production in cleared and thinned stands of semiarid tropical woodland, Brazil. *Forest Ecology and Management,* 23, 201-214.

(2) Reader, R.J., and Bricker, B.D. (1992) Value of selectively cut deciduous forest for understory herb conservation: An experimental assessment. *Forest Ecology and Management,* 51, 317-327.

(3) Pérez, B. and Moreno, J.M. (1998) Fire-type and forestry management effects on the early postfire vegetation dynamics of a *Pinus pinaster* woodland. *Plant Ecology,* 134, 27-41.

(4) Nagaike, T., Kamitani, T., and Nakashizuka, T. (1999) The effect of shelterwood logging on the diversity of plant species in a beech (Fagus crenata) forest in Japan. *Forest Ecology and Management,* 118, 161-171.

(5) Griffis, K.L., Crawford, J.A., Wagner, M.R., and Moir, W.H. (2001) Understory response to management treatments in northern Arizona ponderosa pine forests. *Forest Ecology and Management,* 146, 239-245.

(6) Thysell D.R. and Carey A.B. (2001) Manipulation of density of *Pseudotsuga menziesii* canopies: Preliminary effects on understory vegetation. *Canadian Journal of Forest Research,* 31, 1513-1525.
(7) Sullivan, T.P., Sullivan, D.S., Lindgren, P.M.F., and Boateng, J.O. (2002) Influence of conventional and chemical thinning on stand structure and diversity of plant and mammal communities in young lodgepole pine forest. *Forest Ecology and Management,* 170, 173-187.
(8) Götmark, F., Paltto, H., Nordén, B., and Götmark, E. (2005) Evaluating partial cutting in broadleaved temperate forest under strong experimental control: Short-term effects on herbaceous plants. *Forest Ecology and Management,* 214, 124-141.
(9) Metlen, K.L. and Fiedler, C.E. (2006) Restoration treatment effects on the understory of ponderosa pine/Douglas-fir forests in western Montana, USA. *Forest Ecology and Management,* 222, 355-369.
(10) Montero-Solís, F.M., Sánchez-Velásquez, L.R., Pineda-López, M.D.R.. Martínez-Rivera, L.M., Moermond, T., and Aguirre, J.C. (2006) S Livestock impact on dynamic and structure of tropical dry forest of the Sierra de Manantlán, Mexico. *Journal of Food, Agriculture and Environment*, 4, 266-270.
(11) Powell, G.W., and Bork, E.W. (2006) Aspen canopy removal and root trenching effects on understory vegetation. *Forest Ecology and Management,* 230, 79-90.
(12) Youngblood, A., Metlen, K.L., and Coe, K. (2006) Changes in stand structure and composition after restoration treatments in low elevation dry forests of northeastern Oregon. *Forest Ecology and Management,* 234, 143-163.
(13) Devine, W.E., Harrington, C.A., and Peter, D.H. (2007) Oak woodland restoration: understory response to removal of encroaching conifers. *Ecological Restoration,* 25, 247-255.
(14) Dodson, E.K., Metlen, K.L., and Fiedler, C.E. (2007) Common and uncommon understory species differentially respond to restoration treatments in ponderosa pine/Douglas-fir forests, Montana. *Restoration Ecology,* 15, 696-708.
(15) Schelling, L.R., and McCarthy, B.C. (2007) Effects of prescribed fire and thinning on the spatial heterogeneity of the seed bank in a mixed oak forest. *Natural Areas Journal,* 27, 320-331.
(16) Wayman, R.B., and North, M. (2007) Initial response of a mixed-conifer understory plant community to burning and thinning restoration treatments. *Forest Ecology and Management*, 239, 32-44.
(17) Falk, K.J., Burke, D.M., Elliott, K.A., and Holmes, S.B. (2008) Effects of single-tree and group selection harvesting on the diversity and abundance of spring forest herbs in deciduous forests in southwestern Ontario. *Forest Ecology and Management,* 255, 2486-2494.
(18) Laughlin, D.C., Bakker, J.D., Daniels, M.L., Moore, M.M., Casey, C.A., and Springer, J.D. (2008) Restoring plant species diversity and community composition in a ponderosa pine-bunchgrass ecosystem. *Plant Ecology*, 197, 139-151.
(19) Perchemlides, K.A., Muir, P.S., and Hosten, P.E. (2008) Responses of chaparral and oak woodland plant communities to fuel-reduction thinning in southwestern Oregon. *Rangeland Ecology and Management,* 61, 98-109.
(20) Phillips, R.J., and Waldrop, T.A. (2008) Changes in vegetation structure and composition in response to fuel reduction treatments in the South Carolina Piedmont. *Forest Ecology and Management,* 255, 3107-3116.
(21) Savadogo, P., Tiveau, D., Sawadogo, L., and Tigabu, M. (2008) Herbaceous species responses to long-term effects of prescribed fire, grazing and selective tree cutting in the savanna-woodlands of West Africa. *Perspectives in Plant Ecology, Evolution and Systematics,* 10, 179-195.
(22) Smith, K.J., Keeton, W.S., Twery, M.J., and Tobi, D.R. (2008) Understory plant responses to uneven-aged forestry alternatives in northern hardwood-conifer forests. *Canadian Journal of Forest Research*, 38, 1303-1318.
(23) Waldrop, T.A., Yaussy, D.A., Phillips, R.J., Hutchinson, T. A., Brudnak, L., and Boerner, R.E.J. (2008) Fuel reduction treatments affect stand structure of hardwood forests in Western North Carolina and Southern Ohio, USA. *Forest Ecology and Management,* 225, 3117-3129.
(24) Brockway, D.G., Outcalt, K.W., Estes, B.L., and Rummer, R.B. (2009) Vegetation response to midstorey mulching and prescribed burning for wildfire hazard reduction and longleaf pine (*pinus palustris* mill.) ecosystem restoration. *Forestry,* 82, 299-314.
(25) Sabo, K.E., Sieg, C.H., Hart, S.C., and Bailey, J.D. (2009) The role of disturbance severity and canopy closure on standing crop of understory plant species in ponderosa pine stands in northern Arizona, USA. *Forest Ecology and Management,* 257, 1656-1662.

(26) Wolk, B., and Rocca, M.E. (2009) Thinning and chipping small-diameter ponderosa pine changes understory plant communities on the Colorado Front Range. *Forest Ecology and Management,* 257, 85-95.
(27) Ares, A., Neill, A.R., and Puettmann, K.J. (2010) Understory abundance, species diversity and functional attribute response to thinning in coniferous stands. *Forest Ecology and Management,* 260, 1104-1113.
(28) Brudvig, L.A. (2010) Woody encroachment removal from midwestern oak savannas alters understory diversity across space and time. *Restoration Ecology,* 18, 74-84.
(29) Cole, E.C., Hanley, T.A., and Newton, M. (2013) Influence of precommercial thinning on understory vegetation of young-growth Sitka spruce forests in southeastern Alaska. *Canadian Journal of Forest Research,* 40, 619-628.
(30) Huebner, C.D., Gottschalk, K.W., Miller, G.W., and Brose, P.H. (2010) Restoration of three forest herbs in the Liliaceae family by manipulating deer herbivory and overstorey and understory vegetation. *Plant Ecology and Diversity*, 3, 259-272.
(31) Outcalt, K.W., and Brockway, D.G. (2010) Structure and composition changes following restoration treatments of longleaf pine forests on the Gulf Coastal Plain of Alabama. *Forest Ecology and Management*, 259, 1615-1623.
(32) Lencinas, M.V., Pastur, G.M., Gallo, E., and Cellini, J.M. (2011) Alternative silvicultural practices with variable retention to improve understory plant diversity conservation in southern Patagonia. *Forest Ecology and Management,* 262, 1236-1250.
(33) Stoddard, M.T., McGlone, C.M., Fulé, P.Z., Laughlin, D.C., Daniels, M.L. (2011) Native plants dominate understory vegetation following ponderosa pine forest restoration treatments. *Western North American Naturalist,* 71, 206-214.
(34) Berger, C.A., Puettmann, K.J., and McKenna, J. (2012) Understory Response to Repeated Thinning in Douglas-fir Forests of Western Oregon. *Journal of Sustainable Forestry,* 31, 589-605.
(35) Ross, M.R., Castle, S.C., and Barger, N.N. (2012) Effects of fuels reductions on plant communities and soils in a Piñon-juniper woodland. *Journal of Arid Environments*, 79, 84-92.
(36) Walker, R.F., Fecko, R.M., Frederick, W.B., Miller, W.W., and Johnson, D.W. (2012) Influences of thinning, chipping, and fire on understory vegetation in a Sierran mixed conifer stand. *Journal of Sustainable Forestry,* 31, 493-517.
(37) Huffman, D.W., Stoddard, M.T., Springer, J.D., Crouse, J.E., and Chancellor, W.W. (2013) Understory plant community responses to hazardous fuels reduction treatments in pinyon-juniper woodlands of Arizona, USA. *Forest Ecology and Management*, 289, 478-488.

5.1.4. Thin trees within forests: effects on non-vascular plants

- Four studies (including one replicated, randomized, controlled study) in Canada[3], Finland[1], and Sweden[2,4] examined the effects of thinning trees in forests **on non-vascular plants**. Three found it decreased epiphytic plant abundance[3] and species richness[1,3]. Three found mixed effects depending on thinning method[1] and species[2,4].

A replicated, site comparison study in 1994-1997 in boreal forest in Finland (1) found that **thinning decreased the number epiphytic lichen species**. The total number of epiphytic lichen species/ha was lower in early cut (69) than old growth (88) and similar to both treatments in late cut forest (78). Numbers of epiphytic lichen species/ha occurring on Norway spruce *Picea abies* was lower in early cut (47) than late cut and old growth forest (54-56). Data were collected in 15 sample plots (1 ha) classified according to the age of the dominant tree Norway spruce and recent signs of cutting, there were five replicates of the three treatments: early-cut (age 102 years, 465 cut stumps/ha), late-cut (age 135 years, 247 cut stumps/ha) and old-growth forest (age 161 years, 3 cut stumps/ha).

A replicated, controlled, before-and-after study in 2000–2004 in boreal forest in Sweden (2) found that **thinning increased the number of lichen species but**

also increased the extinction rate of some bryophytes species. Numbers of lichen species/stump increased more in thinned (76%) than in unthinned plots (26%). The increase in number of species/stump for mosses (thinned: 10%; unthinned: 50%) and liverworts (thinned: -50%; unthinned: -10%), and number of species/log for lichens (thinned: 35%; unthinned: 0%), mosses (thinned and unthinned: 30%) and liverworts (thinned: -15%; unthinned: -10%) was similar between treatments. Extinction rate (number of species lost after thinning/total number of species before thinning) for generalist species (living on at least two substrate types) was higher in thinned (43%) than in unthinned plots (16%). Extinction rate was similar between treatments for species living on bark or on both wood and bark (thinned: 75%; unthinned: 65%) and species living on dead wood (thinned and unthinned: 65%). Sites were 15 pairs of thinned (conifers and medium-sized trees removed in October 2002) and unthinned plots (1 ha) situated at least 20 km apart from each other. Data were collected before (September-November 2000) and after (October 2004) thinning.

A replicated, randomized, controlled study in 1998-2004 in boreal forest in Alberta, Canada (3) found **that thinning treatments decreased species richness and abundance of non-vascular plants that grow on other plants**. Numbers of species was the lowest in 10% and 50% canopy retention sites (5/tree), intermediate and similar to the other treatments in the 75% canopy retention sites (6) and highest in unharvested sites (8). The abundance was lower in 10, 50 and 75% canopy retention sites (present in 19, 21 and 25% of sampling points respectively) than in unharvested sites (50% of sampling points). In 2004, six to eight trees were sampled in each of four harvesting treatments (10 ha): 10%, 50% and 75% canopy retention and unharvested, randomly applied in 1998-1999 in each of three sites (a total of 80 trees).

A replicated, controlled, before-and-after study in 2000-2009 in boreal forest in Sweden (4) found that **thinning prevented a decrease in the number of lichen species, but not of mosses**. The change in total number of epiphyte species/plot was negative in unthinned (-3.3) and different than in thinned (1.0), as well as number of lichen species (unthinned: -3.3; thinned: 0.0). The change in the number of bryophytes species was similar (0.1-0.8 species/plot) between treatments. Epiphytes were recorded before and after treatment (2001 and 2009) on five oak trunks in each plot (1 ha). There were 24 pairs of thinned (25% of tree basal area removed in 2002-2003) and unthinned plots.

(1) Kuusinen, M., and Siitonen, J. (1998) Epiphytic lichen diversity in old-growth and managed Picea abies stands in southern Finland. *Journal of Vegetation Science*, 9, 283-292.

(2) Paltto, H., Nordén, B., and Götmark, F. (2008) Partial cutting as a conservation alternative for oak (Quercus spp.) forest-Response of bryophytes and lichens on dead wood. *Forest Ecology and Management,* 256, 536-547.

(3) Caners, R.T., Macdonald, S.E., and Belland, R.J. (2010) Responses of boreal epiphytic bryophytes to different levels of partial canopy harvest. *Botany,* 88, 315-328.

(4) Nordén, B., Paltto, H., Claesson, C., and Götmark, F. (2012) Partial cutting can enhance epiphyte conservation in temperate oak-rich forests. *Forest Ecology and Management,* 270, 35-44.

5.2. Log/remove trees within forests

Background

Here logging is defined as the selective removal of trees with the aim of removing tree biomass. This helps to restore natural open woodland by creating gaps and increasing light availability within the forest, which may increase the growth of the remaining vegetation. Interventions where trees are removed to enhance the future condition of a forest and the development of remaining trees are discussed under 'Thin trees within forests'. Studies comparing the effects of partial logging with clearcutting are discussed in 'Use partial retention harvesting instead of clearcutting'.

5.2.1. Log/remove trees within forests: effects on mature trees

- Three of seven studies (including two replicated, controlled studies) in Bolivia[1], Central African Republic[5], China[12], Finland[4], Malaysia[6], Uganda[13] and the USA[2] found that logging trees in forests decreased the **density and cover of trees**[1,5,13]. Two found it increased tree density[2,12] and two found no effect[4,6] of logging on tree density.
- Three of six studies (including one replicated, randomized, controlled study) in Bolivia[10], Canada[11], China[12], Kenya[7], Malaysia[6] and the USA[8] found that logging trees in forests increased **tree size**[8,10,11]. Two found it decreased tree size[4,12] and one found no effect[7] of logging on tree size.
- Two of four studies (including one paired site study) in Bolivia[15], China[12], Mexico[14] and Papua New Guinea[3] found that logging trees in forests decreased **tree species richness and diversity**[3,14]. One study found it increased diversity[12] and one found no effect[15] of logging on tree species diversity.
- One replicated, controlled study in Canada[9] found that logging trees in forests increased **tree mortality rate**.

A replicated, controlled study in 1996-1999 in tropical forest in Bolivia (1) found that **selective logging decreased trees canopy cover**. Tree canopy cover was higher in unlogged plots (logged: 18%; unlogged: 98%). The number of new commercial tree stems/m^2 was similar in logged (11) and unlogged plots (11). Four logged (single tree selection in 1996 and 1997 on a diameter-limit basis) and four unlogged (control) plots (1 × 1 m) were replicated in nine block over a 200 ha area. Data were collected 14 months after treatment.

A site comparison study in temperate coniferous forest in Colorado, USA (2) found that tree **density was generally higher in logged compared with unlogged area**. Density of trees (trees/ha) over 1.4 m tall was higher in the logged area in east and west facing (logged: 728; unlogged: 350) and flat, high altitude plots (logged: 432; unlogged: 214), but similar between areas in north facing (1153-1229), south facing (219-402) and low altitude riparian plots (379-1148). Total basal area (m^2/ha) was higher in logged flat high altitude plots (logged: 19; unlogged: 11), lower in logged north facing plots (logged: 24; unlogged: 31) and similar between areas in south facing (15-16), east and west facing (18-20) and low altitude riparian plots (21-22). Data were collected in five north facing, five south facing, five east and west facing, five flat high altitude and five low altitude riparian plots (0.1 ha) in each of logged (since the late 1800s) and unlogged (since end of 19th century) study areas (4 km^2).

A site comparison study in 1990-1998 in tropical rain forest in Papua New Guinea (3) found that **high intensity logging decreased tree species diversity**. Species diversity was lowest in high intensity logging (Shannon's index: 0.85) and similar in low intensity logging (1.08) and unlogged plots (1.14). Tree species diversity was calculated in six 0.2 ha plots (200 × 10 m) in each of high intensity logging (20 m^2/ha of basal area removed using conventional high impact logging technics in 1990-1991), low-intensity logging (4.2 m^2/ha of basal area removed using low impact portable sawmill in 1992) and unlogged sites. Data were collected six years after treatments.

A replicated, controlled study in 1984-1996 in temperate coniferous forest in Finland (4) found that **cutting treatments decreased the wood volume but not the number of trees**. Tree volume (m^3/ha) was higher in uncut (248) than in six cutting treatments (32-153). However, there was no difference between treatments for the number of trees/ha (uncut: 2,226; cutting treatments: 1,684-2,669) or seedlings/ha (uncut: 6,156; cutting treatments: 6,109-15,625). Four replicates of seven treatment units (1-3 ha) were established in 1984: uncut; Norway spruce *Picea abies* shelterwood (330 spruce trees left); Scots pine *Pinus sylvestris* shelterwood (220 pine trees left); mixed shelterwood (450 trees left); single-tree selection; group selection (~25 m openings); diameter cutting (>25 cm DBH). Volume and number of trees (>1.3 m tall) were monitored in 1996 in one 40 × 40 m plot in each treatment unit. Seedlings (0.1-1.3 m tall) were monitored in 1991 in 16 subplots (10 m^2) within each plot.

A site comparison study in 2000 in tropical forest in Central African Republic (5) found that **selective logging decreased the density of trees and shrubs over 18 months**. The densities of trees (stems/ha) and shrubs 2.5-10 cm and >10 cm diameter at breast height were lower in 18 years post-logging (trees: 2,212; shrubs: 360) than in 6 months post-logging (trees: 2,806; shrubs: 451) and unlogged treatments (trees: 2,937; shrubs: 451). **Species diversities (Shannon's index) were similar in all treatments** (1.89, 2.00 and 1.94 for, 6-months-post-logging and 18-years-post-logging, respectively) **as well as trees and shrubs basal areas** (unlogged: 30; 6 months post-logging: 30; 18 years post-logging: 24 m^2/ha). Monitoring was in sixteen 30 × 30 m plots in each of three forest sections of different logging histories: unlogged, 6 month and 18 years post-logging (selective logging of timber trees).

A site comparison study in 1958-1997 in tropical rain forest in Malaysia (6) found that **logging decreased tree height and canopy size but not their density**. Unlogged plots had greater canopy height (logged: 24.8 m; unlogged: 27.4 m), canopy surface area (logged: 19,272 m^2/ha; unlogged: 27,845 m^2/ha) and crown size of individual trees (logged: 42.9 m^2; unlogged: 94.5 m^2) compared to logged plots. However, the number of stems/ha was similar between treatments (logged: 6,067; unlogged: 6,418). Data were collected in 1997 using aerial photographs in a 6 ha logged site (all trees >45 cm diameter at breast height removed in 1958) and a 50 ha unlogged site, both divided into 50 × 50 m plots.

A site comparison study in 2000-2003 in tropical moist lower montane forest in Kenya (7) found **no effect of logging on forest structure**. There was no difference between logged and unlogged sites for the maximum height of trees (logged: 25-30 m; unlogged: 26-34 m), height of shrubs (logged: 2.0-2.9 m; unlogged: 2.5 m) and herbaceous layers (logged: 0.4-1.4 m; unlogged: 1.0-1.3 m). Data were collected in 0.5-1.6 ha transects within each of three logged (logged at different time intervals in 1960-1998) and two unlogged sites (no evidence for logging).

A replicated, controlled study in 2003-2005 in temperate coniferous forest in Montana USA (8) found that **selective cutting increased the growth rate of trees.** Tree basal area increase in 10 years was higher in cut (137 cm^2) than in uncut plots (75 cm^2). One cut plot (modified individual tree selection cutting in 1992-1993) and one uncut plot (50 × 50 to 60 × 60 m) was established at each of three sites. Trees were measured in 1992-1993 and again in 2003.

A replicated, controlled study in boreal mixed wood forest in Alberta, Canada (9) found that **harvesting increased tree mortality rate**. Annual mortality was higher in harvested than in unharvested plots for balsam poplar *Populus balsamifera* (harvest: 9.4%; unharvested: 2.3%), paper birch *Betula papyrifera* (harvested: 8.7%; unharvested: 3.1%) and trembling aspen *Populus tremuloides* (harvested: 5.8%; unharvested: 1.7%). Annual mortality of white spruce *Picea glaucae* was similar between treatments (harvested: 2.6%; unharvested: 1.1%). Fifty five harvested (retaining 10% of the trees) and 29 unharvested plots (100 m radius) were established within a 6,900 ha area. Harvesting was in 2000, data were collected annually 2001-2005.

A replicated, randomized, controlled study in 2001-2006 in tropical moist forest in Bolivia (10) found that **increased logging and silvicultural treatment intensity increased tree growth rate**. Tree annual growth rate increased from unlogged (0.32 cm) to normal logging (0.38 cm) to light silviculture (0.42 cm) to intensive silviculture (0.48 cm) treatments. Four treatment plots (27 ha) were randomly established in each of three blocks in 2001-2002: unlogged; normal-logging (regular local logging technics); light silviculture (normal-logging with additional application of low-intensity silvicultural treatments) and intensive silviculture (logged at twice the intensity of the normal-logging treatment with application of intensive silvicultural treatments). Data were collected for four years after treatment.

A replicated, study in 2004 in temperate broadleaf forest in Ontario, Canada (11) found that **selective harvest increased the growth rate of shade-tolerant tree species.** Annual increase of stem diameter (mm) for stems of the shade-tolerant species sugar maple *Acer saccharum* (Before: 1.3; after: 1.4), American beech *Fagus grandifolia* (Before: 1.3; after: 1.7) and eastern hemlock *Tsuga canadensis* (Before: 1.4; after: 1.6) was higher 4-15 years after harvest than in the five years before harvest. In contrast, for the other less shade-tolerant species black cherry *Prunus serotina,* white spruce *Picea glauca,* red maple *Acer rubrum* and yellow birch *Betula alleghaniensis*), stem diameter increase was similar between the two time-periods (1.2-1.6 mm/year). Annual increase of stem diameter was calculated by measuring stem cores extracted in 2004 from 4,127 trees in 174 plots representing nine years of harvest (retaining 15-18 m^2/ha basal area): 1984, 1989, 1992, 1994, 1997, 1998, 2001, 2002, 2003. There were 16-20 plots for each harvest year.

A site comparison study in 2008 in temperate mixed forest in China (12) found that **logging decreased tree size but increased tree density, species richness and diversity 37 years later**. Overall, tree basal area (unlogged: 38

m^2/ha; logged: 27 m^2/ha) and average diameter at breast height (unlogged: 15 cm; logged: 8 cm) were higher in unlogged forest. In contrast, the number of trees/ha (unlogged: 994; logged: 1,921), tree species richness (unlogged: 15 species/0.04 ha; logged: 18 species/0.04 ha) and diversity (Shannon's index unlogged: 3.18; logged: 3.46) were higher in the logged forest. Data were collected in 2008 in four subplots (20 × 20 m) within each of 16 plots (40 × 40 m). Eight were in logged forest (timber harvest of 30% by volume in 1988) and eight in an unlogged primary forest site.

A site comparison study in 2006 in tropical forest in Uganda (13) found that moderate and heavy logging decreased the density of tree stems. Stem density was higher in unlogged and light-logged plots (470 and 480 stems/ha respectively) than in heavy-logged plots (300 stems/ha). There was no difference to other treatments in moderate-logged plots (350 stems/ha). Trees basal area was higher in unlogged (42 m^2/ha) than in moderate-logged and heavy-logged plots (23 m^2/ha in both). There was no difference to other treatments in light-logged plots (33 m^2/ha). Twenty six 200 × 10 m plots were marked in four sites with different logging histories: heavy-logged (n = 5); moderate-logged (n = 4); light-logged (n = 6); and unlogged (n = 11). Logging was in 1969, data were collected in 2006.

A paired-site study in 1996-2006 in tropical moist forest in Mexico (14) found that **logging decreased tree species richness and diversity**. The number of tree species/0.1 ha was higher in unlogged than in logged sites for trees 1-5, 5-10 and 10-25 cm diameter at breast height (268 vs 160, 156 vs 114 and 146 vs 116 respectively) but similar for trees >25 cm (54 vs 41). Species diversity (Shannon's index) was higher in unlogged than in logged sites for trees 1-5 cm diameter at breast height (3.4 vs 2.6) but similar for trees 5-10, 10-25 and >25 cm (2.7 vs 2.3, 2.6 vs 2.4 and 2.3 vs 2.1 respectively). Two pairs of logged (in 1996) and unlogged areas were located at each of three forest sites. Sampling was in 2006 in ten 50 × 20 m transects (total of 0.1 ha) in each of the six logged and six unlogged areas.

A replicated, randomized, controlled, before-and-after study in 2000-2008 in tropical forest in Bolivia (15) found **no effect of logging followed by silviculture treatments on tree species richness or diversity**. There was no difference between before and eight years after treatments in numbers of species/ha (unharvested: 123-122; normal logging: 132-125; light-silviculture: 130-131; intensive-silviculture: 128-130) or diversity (Shannon's index unharvested: -3.06; normal: -0.27; light-silviculture: -0.71; intensive-silviculture: -1.26). Four 27 ha plots were randomly assigned to four treatments: unharvested; normal logging (logging using reduced-impact logging techniques); light-silviculture (logging plus light silviculture); and intensive-silviculture (double logging intensity plus intensive silviculture). Trees were monitored in 2000 and 2008 (before and after treatments) in four 1 ha subplots within each treatment plot.

(1) Fredericksen, T.S., and Mostacedo, B. (2000) Regeneration of timber species following selection logging in a Bolivian tropical dry forest. *Forest Ecology and Management*, 131, 47-55.

(2) Kaufmann, M.R., Regan, C.M., and Brown, P.M. (2000) Heterogeneity in ponderosa pine/Douglas-fir forests: age and size structure in unlogged and logged landscapes of central Colorado. *Canadian Journal of Forest Research,* 30, 698-711.

(3) Lindemalm, F., and Rogers, H.M. (2001) Impacts of conventional logging and portable sawmill logging operations on tree diversity in east New Britain, Papua New Guinea. *Australian Forestry,* 64, 26-31.
(4) Lähde, E., Eskelinen, T., and Väänänen, A. (2002) Growth and diversity effects of silvicultural alternatives on an old-growth forest in Finland. *Forestry,* 75, 395-400.
(5) Hall, J.S., Harris, D.J., Medjibe, V., and Ashton, P.M.S. (2003) The effects of selective logging on forest structure and tree species composition in a Central African forest: Implications for management of conservation areas. *Forest Ecology and Management,* 183, 249-264.
(6) Okuda, T., Suzuki, M., Adachi, N., Quah, E.S., Hussein, N.A., and Manokaran, N. (2003) Effect of selective logging on canopy and stand structure and tree species composition in a lowland dipterocarp forest in peninsular Malaysia. *Forest Ecology and Management,* 157, 294-320.
(7) Hitimana, J., Kiyiapi, J.L., and Njunge, J.T. (2004) Forest structure characteristics in disturbed and undisturbed sites of Mt. Elgon Moist Lower Montane Forest, western Kenya. *Forest Ecology and Management*, 194, 269-291.
(8) Fajardo, A., Graham ,J.M., Goodburn, J.M., and Fiedler, C.E. (2007) Ten-year responses of ponderosa pine growth, vigor, and recruitment to restoration treatments in the Bitterroot Mountains, Montana. USA, *Forest Ecology and Management,* 243, 50-60.
(9) Bladon, K.D., Lieffers, V.J., Silins, U., Landhäusser, S.M., and Blenis, P.V. (2008) Elevated mortality of residual trees following structural retention harvesting in boreal mixedwoods. *Forestry Chronicle,* 84, 70-75.
(10) Peña-Claros, M., Fredericksen, T.S., Alarcón, A., Blate, G.M., Choque, U., Leaño, C., Licona, J.C., Mostacedo, B., Pariona, W., Villegas, Z., and Putz, F.E. (2009) Beyond reduced-impact logging: Silvicultural treatments to increase growth rates of tropical trees. *Forest Ecology and Management*, 256, 1458-1467.
(11) Jones, T.A., Domke, G.M., and Thomas, S.C. (2009) Canopy tree growth responses following selection harvest in seven species varying in shade tolerance. *Canadian Journal of Forest Research*, 39, 430-440.
(12) Su, D., Yu, D., Zhou, L.I., Xie, X., Liu, Z., and Dai, L. (2010) Differences in the structure, species composition and diversity of primary and harvested forests on Changbai Mountain, Northeast China. *Journal of Forest Science*, 56, 285-293.
(13) Bonnell, T.R., Reyna-Hurtado, R., and Chapman, C.A. (2011) Post-logging recovery time is longer than expected in an East African tropical forest. *Forest Ecology and Management,* 261, 855-864.
(14) Gutiérrez-Granados, G., Pérez-Salicrup, D.R., and Dirzo, R. (2011) Differential diameter-size effects of forest management on tree species richness and community structure: Implications for conservation. *Biodiversity and Conservation,* 20, 1571-1585.
(15) Carreño-Rocabado, G., Peña-Claros, M., Bongers, F., Alarcón, A., Licona, J-C., and Poorter, L. (2012) Effects of disturbance intensity on species and functional diversity in a tropical forest. *Journal of Ecology,* 100, 1453-1463.

5.2.2. Log/remove trees within forests: effects on young trees

- One replicated controlled study in Canada[1] found that logging trees in forests increased the **density of young trees**[1]. One replicated controlled study in Costa Rica[2] found mixed effects[2] on the density of young trees.

A replicated, controlled study in 1992-2001 in boreal forest in Ontario, Canada (1) found that **structural retention harvest increased tree sapling density**. Average sapling density increased from 4,178 to 5,109 saplings/ha in harvested compared with unharvested plots. Harvesting was carried out in 1992. Remaining trees were healthy seed bearers and declining quality trees. Six unharvested control plots and 12 harvested plots, spread over an area of approximately 1,200 ha were monitored during August and September 2001. Plot areas varied from 3 to 104 ha (average 26 ha). Fifty five sample points were placed within control plots and 89 within harvested plots (3–20 points/plot). Tree saplings were recorded inside a 5 m radius ring around plot centre.

A replicated, controlled study in 1997-2002 in tropical rain forest in Costa Rica (2) found that **selective logging decreased the density of seedlings and small juvenile trees but increased the number of larger trees**. For *Caryocar costaricense*, the density (individuals/ha) of seedlings (<50 cm tall) (logged: 3.1; unlogged: 4.5) and small juveniles (<2 cm diameter at breast height) (logged: 5.2; unlogged: 8.0) was higher in unlogged plots. In contrast, the density of large juveniles (2-10 cm diameter at breast height) was higher in logged plots (logged: 4.3; unlogged: 2.4). For purpleheart *Peltogyne purpurea*, the density of seedlings (logged: 208.8; unlogged: 511.2) was higher in unlogged plots, while the density did not differe for small (logged: 2.2; unlogged: 3.1) and large juveniles (logged: 2.6; unlogged: 2.2). Data were collected in 2002 in three logged (selective logging in 1997-1998) and three unlogged plots (100 × 30 m) in each of 11 sites.

(1) Bebber, D.P., Cole, W.G., Thomas, S.C., Balsillie, D., and Duinker, P. (2005) Effects of retention harvests on structure of old-growth Pinus strobus L. stands in Ontario. *Forest Ecology and Management,* 205, 91-103.

(2) Lobo, J., Barrantes, G., Castillo, M., Quesada, R., Maldonado, T., Fuchs, E.J., Solís, S., and Quesada, M. (2007) Effects of selective logging on the abundance, regeneration and short-term survival of *Caryocar costaricense* (Caryocaceae) and Peltogyne purpurea (Caesalpinaceae), two endemic timber species of southern Central America. *Forest Ecology and Management*, 245, 88-95.

5.2.3. Log/remove trees within forests: effects on understory plants

- Five of 10 studies (including four replicated, randomized, controlled studies) in Bolivia[2], Canada[3,8,11,12], India[6] and the USA[1,4,7,9] found that logging trees in forests increased the **density and cover of understory plants**[1,2,7,8,12]. Five studies found no effect[4,9,11] or mixed effects[3,6].
- Four of seven studies (including one replicated, randomized, controlled study) in Australia[5], Canada[3,10,11,12] and the USA[4,9] found that logging trees in forests increased **species richness and diversity of understory** plants[4,5,10,12]. Three studies found no effect[3,9,11].

A replicated, randomized, controlled study in 1991-1993 in temperate coniferous forest in Oregon USA (1) found that **cutting western juniper *Juniperus occidentalis* trees increased total biomass and cover of understory perennial plants.** The total biomass of understory perennial plants (cut: 329 kg/ha; uncut: 38 kg/ha) and their cover (cut: 4.3-4.8%; uncut: 1.4-1.5%) was higher in cut plots. In 1993, total biomass was sampled at 3 m intervals with 1 m^2 quadrats along two 45 m transects. Cover of perennial plants was measured along five 30.5 m line transects in each plot of eight pairs of cut (all juniper trees were cut down in 1991) and uncut 0.4 ha plots.

A replicated, controlled study in 1996-1999 in tropical forest in Bolivia (2) found that **selective logging increased ground vegetation cover**. Ground vegetation cover was higher in logged (99%) than unlogged plots (81%). Four logged (single tree selection in 1996 and 1997 on a diameter-limit basis) and four unlogged plots (1 × 1 m) were replicated in nine block over a 200 ha area. Data were collected 14 months after treatment.

A replicated, controlled study in 1988-2001 in temperate coniferous forest in British Columbia, Canada (3) found that **harvesting had a mixed effect on understory plant cover, but did not affect species richness or diversity**. The cover of the most common herbaceous species bluejoint reedgrass *Calamagrostis canadensis* (harvested: 27.7%-38.9%; unharvested: 9.7%-17.9%) as well as of four more herbaceous species (harvested: 1.3%-5.9%; unharvested: 0.5%-3.7%) was higher in harvested than unharvested plots. In contrast, the cover of Mountain Sweet Cicely *Osmorhiza berteroi* (harvested: 0.0%; unharvested: 0.2%-0.7%) was higher in unharvested plots. Cover of regenerating trees was higher in harvested plots: trembling aspen *Populus tremuloides* (harvested: 54.9%-63.9%; unharvested: 3.5%-4.1%) and balsam poplar *Populus balsamifera* (harvested: 8.6%-12.5%; unharvested: 0.0%-0.1%) However, the cover of the shrub birch-leaved spirea *Spiraea betulifolia* was higher in unharvested plots (harvested: 0.7%-1.0%; unharvested: 7.7%). Numbers of species/5 m^2 (harvested: 38-41; unharvested: 34-39) and plant diversity (Shannon's index harvested: 2.53-2.74; unharvested: 2.78-2.89) were similar between treatments. Data were collected in 2001 in three replicates of two harvested (in 1988-1989) and two unharvested mature aspen plots (5 ha). Species richness and diversity were calculated for 40 subplots of 0.125 m^2 in each plot (total of 5 m^2).

A replicated, controlled study in 1951-1998 in coniferous montane and subalpine forest in Wyoming, USA (4) found that **harvesting increased species richness but not cover of understory plants**. The number of species/forest unit was higher in harvested than control plots in both montane (harvested: 26; unharvested: 19) and subalpine forest units (harvested: 32; unharvested: 12). Total cover of understory plants was similar between treatments in both montane (harvested: 25%; unharvested: 23%) and subalpine units (harvested: 58%; control: 55%). Data were collected in 1997-1998 using 50 frames (50 × 100 cm) at each of 30 harvested (in 1951-1969) and 24 unharvested forest units (< 0.5 ha).

A replicated, before-and-after study in 1987-1993 in wet sclerophyll eucalypt forest in Tasmania, Australia (5) found that **cutting treatments increased plant species richness.** All treatments increased species richness/plot: clearcutting (before: 11; after: 15), group-selection (before: 13; after: 18) and partial-logging (before: 15; after: 22). Data were collected before (1987) and after (1995-1996) treatments in 44 group-selection (100 m diameter clearcut gaps) and 103 partial-logging (retaining 25%-50% of stems) plots (5 × 5 m), and in 25 clearcutting plots (20 × 20 m).

A site comparison study in 2000-2001 in tropical moist lowland forest in India (6) found that **selective logging had a mixed effect on the abundance of ferns and other epiphytic plants**. Abundance (individuals/25 × 25 m plot) of **ferns** and non-orchid epiphytes were lower in logged (28 and 33 respectively) than in unlogged plots (121-128 and 170-208 respectively). Abundance of epiphytic orchids was similar between treatments (35 vs 28-44). In 2000-2001, non-orchid epiphytes were monitored in four logged (selective logging 1960-1996) and eight unlogged plots (25× 25 m).

A replicated, randomized, controlled study in 1999-2001 in temperate coniferous forest in Oregon, USA (7) found that **cutting western juniper** ***Juniperus occidentalis*** **trees increased total herbaceous cover and seed production of perennial grasses**. Herbaceous plant cover in cut plots (16%) was higher than in uncut plots (4%). Seed production of perennial grasses was higher in cut plots (42 kg/ha) than in uncut plots (<1 kg/ha), while seed production of Sandberg's bluegrass *Poa sandbergii* was similar (5 kg/ha) in both treatments. In 2001, herbaceous cover was estimated using 0.2 m^2 frames and seed production was estimated using five 9 m^2 frames in four pairs of cut (all juniper trees cut down in 1998) and uncut plots (0.45 ha).

A replicated, controlled study in 1992-2001 in boreal forest in Ontario, Canada (8) found that **structural retention harvest increased herbaceous vegetation cover**. Average herbaceous vegetation cover was 40% in harvested compared to 26% in unharvested plots. Harvesting was carried out in 1992. Residual trees were healthy seed bearers and declining quality trees. Six unharvested control plots and 12 harvested plots, spread over an area of approximately 1,200 ha were monitored during August and September 2001. Plot areas varied from 3 to 104 ha (average 26 ha). Fifty five sample points were placed within control plots and 89 within harvested plots (3–20 points/plot). Herbaceous vegetation was recorded inside a 5 m radius ring around plot centre.

A replicated, randomized, controlled study in 1952-1991 in temperate broadleaf forest in Wisconsin, USA (9) found **no effect of cutting on ground layer plant species richness and diversity**. For spring and summer flowering plants, species richness (1-6 species/150 m^2 and 1-18 species/1 m^2 respectively) and diversity (Shannon's Index 0.57 and 0.71 respectively) were similar between treatments. Six treatments (1ha): diameter-limit cut (5.3 m^2/ha residual basal area, applied in 1952); shelterwood cut (9.2 m^2/ha residual basal area, applied in 1957); three levels of individual tree selection: light (20.6 m^2/ha residual basal area), medium (17.2 m^2/ha residual basal area) and heavy (13.8 m^2/ha residual basal area), applied in 1952, 1962, 1972 and 1982; and uncut, were randomly replicated in three blocks. In 1991, spring ephemeral species were monitored in five 10 × 15 m plots and summer flowering species in eight 1 m^2 plots in each treatment.

A replicated, controlled study in 1968-2002 in temperate mixed wood forest in Alberta, Canada (10) found that **salvage logging increased the number of shrub species in early successional forest**. In early successional forest, shrub species richness/100 m^2 was higher in logged plots (logged: 11; unlogged: 9). There was no difference between treatments for herbaceous species richness (logged: 22; unlogged: 18) or all plants (logged: 33; unlogged: 27), or understory plant cover (logged: 94%; unlogged: 108%). In mid-successional forest, species richness/100 m^2 of shrubs (logged: 20; unlogged: 18), herbs (logged: 26; unlogged: 23) and of all plants (logged: 22; unlogged: 41), and understory plant cover (logged: 88%; unlogged: 99%) were similar between treatments. Data were collected in 2002 in five logged (common operational salvage-logging) and five unlogged forest units. Two logged and two unlogged plots were established in mid-successional forests (burned by wildfire in 1968) and the other six in early successional forests (burned by wildfire in 1999). Understory plant cover was evaluated in two plots (1 m^2) and species richness was determined in one plot (100 m^2) in each of 13-20 sites within each forest unit.

A replicated, randomized, controlled study in 1993-1996 in boreal forest in Manitoba, Canada (11) found **no effect of cutting on plant cover and diversity**. Total plant cover (uncut: 89-132%; harvest to stump: 78-107%; full tree removal: 76-103%), and plant species richness/2 m^2 plot (uncut: 12-19; harvested to stump: 12-18; full tree removal: 12-20) and diversity (Simpson's index uncut: 3.9-6.0; harvested to stump: 4.4-3.0; full tree removal: 4.5-6.3) were similar between treatments. In 1993, three plots (30 × 100 m) of each uncut, harvested to stump and full tree removal (harvested trees completely removed) treatments were randomly applied in each of six blocks. Plant cover was measured in six subplots (5 × 5 m) within each plot (total of 324 subplots). Species richness and diversity were determined in a 1 × 2 m quadrat in each subplot. Data were collected in 1996.

A before-and-after trial in 1992-2003 in boreal forest in Quebec, Canada (12) found that **conifer cutting increased understory species richness, diversity and cover**. Numbers of species/1 m^2 plot increased in conifer cut plots (before: 4-9; after: 7-13) and remained similar in uncut plots (before: 5-10; after: 6-12). Species diversity (Shannon's index) increased in conifer cut plots (before: 0.7-1.3; after: 0.9-1.5) and remained similar in uncut plots (before: 0.7-1.3; after: 0.8-1.5). Cover increased in conifer cut plots (before: 70%-80%; after: 100%-170%) and remained similar in uncut plots (before: 90%-100%; after: 90%-120%). In 1992, confer cutting (all conifers cut and removed) and uncut treatments (100 m^2) were replicated in three blocks (>625 m^2) at each of two sites. Data were collected before (1992) and after treatments (2003) in 5-12 plots (1 m^2) in each treatment.

(1) Bates, J.D., Miller, R.F., and Svejcar, T.J. (2000) Understory dynamics in cut and uncut western juniper woodlands. Journal of Range Management, 53, 119-126.

(2) Fredericksen, T.S., and Mostacedo, B. (2000) Regeneration of timber species following selection logging in a Bolivian tropical dry forest. *Forest Ecology and Management*, 131, 47-55.

(3) Krzic, M., Newman, R.F., and Broersma, K. (2003) Plant species diversity and soil quality in harvested and grazed boreal aspen stands of northeastern British Columbia. *Forest Ecology and Management,* 182, 315-325.

(4) Selmants, P.C., and Knight, D.H. (2003) Understory plant species composition 30-50 years after clearcutting in southeastern Wyoming coniferous forests. *Forest Ecology and Management,* 185, 275-289.

(5) Wapstra, M., Duncan, F., Williams, K., and Walsh, D. (2003) Effect of silvicultural system on vascular flora in a wet sclerophyll forest in south-eastern Tasmania. *Australian Forestry,* 66, 247-257.

(6) Padmawathe, R., Qureshi, Q., and Rawat, G.S. (2004) Effects of selective logging on vascular epiphyte diversity in a moist lowland forest of Eastern Himalaya, India. *Biological Conservation*, 119, 81-89.

(7) Bates, J.D. (2005) Herbaceous response to cattle grazing following juniper cutting in Oregon. *Rangeland Ecology and Management,* 58, 225-233.

(8) Bebber, D.P., Cole, W.G., Thomas, S.C., Balsillie, D., and Duinker, P. (2005) Effects of retention harvests on structure of old-growth Pinus strobus L. stands in Ontario. *Forest Ecology and Management,* 205, 91-103.

(9) Kern, C.C., Palik, B.J., and Strong, T.F. (2006) Ground-layer plant community responses to even-age and uneven-age silvicultural treatments in Wisconsin northern hardwood forests. *Forest Ecology and Management,* 230, 162-170.

(10) Kurulok, S. E., and Macdonald, S.E. (2007) Impacts of postfire salvage logging on understory plant communities of the boreal mixedwood forest 2 and 34 years after disturbance. *Canadian Journal of Forest Research,* 37, 2637-2651.

(11) Kembel, S.W., Waters, I., and Shay, J.M. (2008) Short-term effects of cut-to-length versus full-tree harvesting on understory plant communities and understory-regeneration associations in Manitoba boreal forests. *Forest Ecology and Management,* 255, 1848-1858.
(12) Grandpré, L., Boucher, D., Bergeron, Y., and Gagnon, D. (2011) Effects of small canopy gaps on boreal mixedwood understory vegetation dynamics. *Community Ecology*, 12, 67-77.

5.2.4. Log/remove trees within forests: effects on non-vascular plants

- Two of three studies (including one replicated, paired sites study) in Australia[1], Norway[3] and Sweden[3] found logging trees in forests decreased epiphytic plant abundance[2] and fern fertility[1]. One found mixed effects depending on species[3].

A site comparison study in 1991-1994 in wet eucalyptus forest in Victoria, Australia (1) found that **logging trees decreased the percentage of fertile tree ferns and the number of living leaves, but not the number of leaves produced**. The percentage of fertile ferns (thinned: 30-31%; control: 86-89%) and the number of living leaves/fern (thinned: 2-11; control: 22-29) was higher in control sites, while the annual number of leaves/fern produced was similar between sites (thinned: 10; control: 14-18). Two tree ferns, soft tree fern *Dicksonia antarctica* and rough tree fern *Cyathea australis,* were monitored in five 30 × 30 m plots in a 12 ha thinned site (logged in 1991-1992). An additional 51 soft tree fern and nine rough tree fern individuals were monitored in unlogged sites. Data were collected two years after thinning.

A controlled study in 1995-2001 in boreal forest in Norway (2) found that **logging decreased cover and abundance of lichens**. For *Cavernularia hultenii*, cover and abundance (number of lichen branches/m branch length) were lower in sites that were thinned by cutting few relatively large gaps (cover: 2.4%; abundance: 2.6) than in sites that were thinned by cutting a large number of relatively small gaps (cover: 4.2%; abundance: 5.5). Cover and abundance were the highest in unthinned sites (cover: 6.2%; abundance: 8.4). For *Platismatia glauca* cover (large gaps: 22.6%; small gaps: 30.4%; unthinned: 29.1%) and abundance (large gaps: 9.1; small gaps: 13.3; unthinned: 13.8) were lower in large gaps sites than in small gaps and unthinned sites. For Norwegian ragged lichen *Platismatia norvegica* cover (3.0-3.7%) and abundance (0.6-0.9) were similar between treatments. A 100 ha area was divided into large gaps (three clearcuts of 150 × 150 m), small gaps (23 clearcuts of 50 × 50 m) and unthinned sections. Logging was applied in 1995-1996. Lichens were monitored in 2001 on 110 trees (>40 cm diameter at breast height): 45 in each logging treatment and 20 in the unlogged section.

A replicated, paired sites study in 1997-2001 in boreal forest in Sweden (3) found that **logging decreased the number of liverwort and increased the number of moss species**. Numbers of liverwort species/plot (0.1 ha) was lower in thinned plots than in uncut, both in the short-term (cut: 25; uncut: 33) and long-term (cut: 27; uncut: 32). Numbers of moss species/plot was higher in cut than in uncut plots in the short-term (cut: 53; uncut: 47) and similar in the long-term (48). Total number of bryophytes species/plot was similar in both short-term (cut: 78; uncut: 80) and long-term (cut: 75; uncut: 80). Liverworts and mosses were monitored in 2001 in 15 short-term (cut in 1998) and 18 long-term (cut in 1950-1970) pairs of cut and uncut 20 × 50 m plots.

(1) Ough, K., and Murphy, A. (1996) The effect of clearfell logging on tree-ferns in Victorian Wet Forest. *Australian Forestry*, 59, 178-188.
(2) Hilmo, O., Hytteborn, H., and Holien, H. (2005) Do different logging strategies influence the abundance of epiphytic chlorolichens? *The Lichenologist*, 37, 543-553.
(3) Dynesius, M., and Hylander, K. (2007) Resilience of bryophyte communities to clear-cutting of boreal stream-side forests. *Biological Conservation*, 135, 423-434.

5.3. Remove woody debris after timber harvesting

- One of six studies (including two replicated, randomized, controlled studies) in the USA[3,8,9,11,12] and France[10] found that woody debris removal increased **understory vegetation cover**[12]. Three studies found no effect[11] or mixed effects[3,8] on cover. Four of the studies[3,8,9,10] found no effect or mixed effects on understory vegetation species richness and diversity and one found no effect of woody debris removal species diversity[10] of trees.
- Six studies (including two replicated, randomized, controlled studies) in Canada[4,7,5], Ethiopia[1], Spain[2] and the USA[6] examined the effect of woody debris removal on **young trees**. One study found that debris removal increased young tree density[4], another study found that it decreased young tree density[6], and three studies found mixed effects[1] or no effect[7,2] on young tree density. One[5] found no effect of woody-debris removal on young tree survival.

Background
Coarse woody debris consists of fallen dead trees and cut branches (> 10 cm diameter) that are left during tree harvesting. Removal of coarse woody debris uncovers the ground and allows sunlight to reach it, which may enhance seed germination and increase plant biodiversity.

A replicated, controlled study in 1992 in Afro-montane forests in Ethiopia (1) found that **woody debris treatments had mixed effects on seedling establishment** of African Juniper *Juniperus procera* and East African yellowwood *Afrocarpus gracilior* trees. Seedling density (individuals/m^2) of African juniper was higher in burned than control and similar to both in raked plots (control: 0-5; raked: 8-12; burned: 13-14), while seedling density of East African yellowwood was lower in burned than control and raked plots (control: 4; raked: 5; burned: 1-3). Data were collected in December 1992 in three plots (10 × 10 m) of each treatment: control, raked (all logging waste and ground vegetation removed, seedbed raked) and burned (logging waste, ground vegetation and litter burned). Plots were established in a 40 × 40 m study site in March-April 1992.

A controlled study in 1995-1998 in temperate coniferous forest in Spain (2) found **no effect of burnt wood removal on the emergence and mortality of Aleppo pine *Pinus halepensis* seedlings**. Emergence rates were similar between treatments (cleared: 0.0-3.2%; control: 0.0-2.6%) and mortality (cleared: 3-18%; control: 2-9%). In June 1995, two treatment plots (2,500 m^2), one cleared (all burnt pines cut down and removed) and one control (untreated) were established in an area that was burnt in August 1994. Seedlings were sampled in 20 plots (4 × 5 m^2) in each treatment plot on six dates during the first three post-treatment years: October 1995, January 1996, June 1996, January 1997, June 1997 and June 1998.

A replicated, controlled study in 1988-1991 in temperate coniferous forest in Washington State, USA (3) found that different **woody debris removal treatments had mixed effects on understory vegetation cover and no effect on species richness**. At one site, vegetation cover was higher in control than other treatments (chopped: 1.8%; spring burn: 2.5%; pulled off site: 4.2%; control: 7.1%). At a second site, cover was higher in control, pulled off and autumn burn treatments (2.9, 1.2 and 1.2% respectively) than spring burn and chopped treatments (0.2% in both). At the other two sites it was similar among treatments (chopped: 2.7-2.8%; spring burn: 2.9-5.7%; autumn burn: 3.8-4.7%; pulled off: 1.2-5.7%; control: 2.1-2.2%). The number of species/m^2 was similar among treatments at all four sites (chopped: 7-26; spring burn: 7-22; autumn burn: 8-20; pulled off: 5-20; control: 10-18). In 1989, five treatment plots (0.25-3.2 ha) were established in each of four sites: control (untreated); pulled off (woody debris pulled off the site); chopped (debris chopped); spring burn (low intensity burn); autumn burn (low to medium intensity). All plots were clearcut in 1988. Data were collected in 1991 in 15 quadrats (1 m^2) in each treatment plot.

A replicated, randomized study in 1995-2000 in boreal forest in British Columbia, Canada (4) found that **woody debris removal treatments increased tree sapling density and decreased their height**. Trembling aspen *Populus tremuloides* sapling density was higher in plots were all parts of the trees removed (tree removal) (44,000 stems/ha) than in plots were only saleable stems removed (stem removal) (34,000). The saplings dominant height was higher in stem removal and tree removal plots (225 and 245 cm respectively) than in plots were the hole forest floor was removed in addition (complete removal) (120 cm). White spruce *Picea glauca* total height was higher in stem removal plots (71 cm) than in tree removal and complete removal plots (54 and 42 cm respectively). Trembling aspen density was monitored in nine stem removal, nine tree removal and nine complete removal 40 × 70 m treatment plots. The height of more than 12 aspen saplings and of 200 randomly selected white spruce saplings was measured in each plot. Treatments were applied in 1995, Data were collected in 2000.

A replicated, controlled study in 2000-2003 in temperate coniferous forest in Québec, Canada (5) found **no effect of woody debris removal and raking in artificial gaps on the survival of yellow birch *Betula alleghaniens* seedlings**. Seedling survival was similar between treatments (debris removal and raking: 45-50%; removal: 22-40%; control: 23-38%). Data were collected in 2003 in six control, six debris removal (mechanically pushing all debris to the edges of the gap), and six removal and raking (pushing all debris followed by raking) artificial forest gaps (900 m^2). Gaps were created and treatments applied in 2000.

A replicated study in 1994-2003 in temperate broadleaf forest in Missouri, USA (6) found that **after wood harvest, removal of the whole tree decreased the density but not the height of young trees compared with removal of main stems only, or removal of the whole tree plus debris from the forest floor**. The number of individuals/m^2 plot for trees was lower in whole tree removal plots (3.9) than in main stem removal plots (4.6) and forest floor debris removal plots (4.6). For shrubs (main stem removal: 2.8; whole tree removal: 3.2; forest floor debris removal: 3.3), woody vines (main stem removal: 4.9; whole tree removal: 3.7; forest floor debris removal: 3.5) and herbs (main stem removal: 7.7; whole tree removal: 7.7; forest floor debris removal: 8.9) numbers of individuals was similar between treatments. Height (m) of trees (main stem removal: 2.6; whole tree removal: 2.6; forest floor debris removal: 2.4) and of all other plants (main stem removal: 0.6; whole tree removal: 0.5; forest floor debris removal: 0.5) was similar between treatments. Data were collected in 2003 in three plots (8 m^2) in each of three replicate treatment plots (0.4 ha): main stem removal, whole tree removal and forest floor debris removal. Harvest and removal treatments were applied in 1994.

A replicated, randomized, controlled study in 2001-2006 in temperate coniferous forest in Alberta, Canada (7) found **no effect of woody debris removal on the density and height of pine seedlings**. The density (1,308 seedlings/ha) and height (20 cm) of seedlings were similar between treatments. Twelve removed (woody debris removal in winter 2001) and 12 unremoved plots (30 × 30 m) were established in 2002. Density and height of regenerated seedlings were measured in 2006 in five subplots (10 m^2) within each plot.

A replicated, controlled study in 2003-2006 in temperate coniferous forest in Colorado USA (8) found that woody **debris removal treatments had mixed effects on plant cover and species richness**. Six to 18 months after treatment, percentage cover and species richness/m^2 of plants were higher in untreated plots and those where debris was cut up and left (46-50% cover, 7 species) than where debris was piled and burned (1% cover, <1 species). After 2.5-3.5 years the percentage cover and species richness/m^2 of plants were highest where debris was cut up (46% cover, 8 species), lower in untreated plots (26%-29% cover, 6 species) and the lowest where debris was piled and burned (4% cover, <1 species). Three treatments were applied in three sites (1-2 km^2): untreated, piled and burned (cutting trees, piling debris and burning in areas 3–6 m^2) and cutting and leaving mulched material (areas 10–12 m^2). Monitoring was in a total of 75 untreated, 50 piled and burned and 50 cut up treatment plots (1 × 1 m).

A replicated, randomized, controlled study in 1999-2006 in temperate coniferous forest in Washington State, USA (9) found **no effect of removing all woody material after clearcutting on plant species richness and diversity compared with removing only tree trunks**. Species richness (trunk removal: 17; complete-removal: 16) and diversity (Simpson's index: trunk removal: 0.36; complete removal: 0.27) were similar between treatments. Data were collected in 2006 in two plots (30 × 85 m) of each treatment, trunk removal only and removal of all woody material. Treatments applied after clearcutting in 1999 in each of four blocks. In all plots Douglas-fir *Pseudotsuga menziesii* seedlings were planted in 2000 and vegetation-control herbicide was applied annually.

A replicated, controlled study in 2005-2008 in temperate forest in France (10) found **no effect of clearing of woody debris on species richness and diversity of trees and herbs**. Numbers of woody plant species (control: 7-8; cleared: 10 m^2) and diversity (Shannon's index control: 2.1-2.5; cleared: 1.9-2.1), and number of herbaceous species (control: 17-20; cleared: 13-17 m^2) and diversity (Shannon's index control: 2.9-3.5; cleared: 3.1-3.4) were similar between treatments. Data were collected in May 2008 in 60 pairs of control (woody debris left) and 60 cleared (woody debris cleared of) plots (1 m^2) in one site, and 42 similar pairs at a second site. Plots were set up in May 2005.

A replicated, randomized, controlled study in 1999-2006 in temperate coniferous forest in Washington State and Oregon, USA (11) found **no effect of different woody debris removal treatments after clearcutting on cover of Douglas-fir *Pseudotsuga menziesii* and understory vegetation**. At all three sites (respectively), cover was similar between treatments for Douglas-fir (control: 13%, 13% and 62%; piled/removed: 11%, 15% and 70%) and understory vegetation (control: 92%, 118% and 73%; piled/removed: 81%, 117% and 5%). Four blocks of 8-16 plots where woody debris had been piled or removed after clearcutting and 8-16 control treatment plots (0.26 ha) were established in each of three sites, all clearcut and planted with Douglas-fir in 1999-2003. Data were collected five years after clearcutting.

A controlled study in 2007-2009 in Piñon-juniper woodland in Utah, USA (12) found that **shredding woody debris (mulching) increased understory vegetation cover**. Cover of understory plants was higher in mulched (66%) than control plots (4%). Two treatment sites (0.4-0.9 km^2) were established in 2007: mulching (using a tractor with an attached brush-cutter) and control (untreated). Data were collected in 2009 along 10 transects (35 m) in each site.

(1) Sharew, H., Legg, C.J., and Grace, J. (1997) Effects of ground preparation and microenvironment on germination and natural regeneration of *Juniperus procera* and *Afrocarpus gracilior* in Ethiopia. *Forest Ecology and Management*, 93, 215-225.

(2) Martínez-Sánchez, J.J., Ferrandis, P., De Las Heras, J., and María Herranz, J. (1999) Effect of burnt wood removal on the natural regeneration of Pinus halepensis after fire in a pine forest in Tus Valley (SE Spain). *Forest Ecology and Management,* 123, 1-10.

(3) Scherer, G., Zabowski, D., Java, B., and Everett, R. (2000) Timber harvesting residue treatment. Part II. Understory vegetation response. *Forest Ecology and Management,* 126, 35-50.

(4) Kabzems, R. and Haeussler, S. (2005) Soil properties, aspen, and white spruce responses 5 years after organic matter removal and compaction treatments. *Canadian Journal of Forest Research,* 35, 2045-2055.

(5) Lorenzetti, F., Delagrange, S., Bouffard, D., and Nolet, P. (2008) Establishment, survivorship, and growth of yellow birch seedlings after site preparation treatments in large gaps. *Forest Ecology and Management,* 254, 350-361.

(6) Ponder Jr., F. (2008) Nine-year response of hardwood understory to organic matter removal and soil compaction. *Northern Journal of Applied Forestry,* 25, 25-31.

(7) Landhäusser, S.M. (2009) Impact of slash removal, drag scarification, and mounding on lodgepole pine cone distribution and seedling regeneration after cut-to-length harvesting on high elevation sites. *Forest Ecology and Management*, 258, 43-49.

(8) Owen, S.M., Sieg, C H., Gehring, C.A., and Bowker, M.A. (2009) Above and belowground responses to tree thinning depend on the treatment of tree debris. *Forest Ecology and Management*, 259, 71-80.

(9) Peter, D.H., and Harrington, C. (2009) Six years of plant community development after clearcut harvesting in western Washington. *Canadian Journal of Forest Research,* 39, 308-319.

(10) Pellerin, M., Saïd, S., Richard, E., Hamann, J-L., Dubois-Coli, C., and Hum, P. (2010) Impact of deer on temperate forest vegetation and woody debris as protection of forest regeneration against browsing. *Forest Ecology and Management,* 260, 429-437.

(11) Peter, D.H., and Harrington, T.B. (2012) Relations of native and exotic species 5 years after clearcutting with and without herbicide and logging debris treatments. *USDA Forest Service* - Research Paper PNW-RP-53.

(12) Ross, M.R., Castle, S.C., and Barger, N.N. (2012) Effects of fuels reductions on plant communities and soils in a Piñon-juniper woodland. *Journal of Arid Environments*, 79, 84-92.

5.4. Use shelterwood harvesting instead of clearcutting

- Three replicated, controlled studies in Sweden[1,3] and the USA[2] found that shelterwood harvesting resulted in higher **plant diversity**[1], lower **grass cover**[3] and higher **density of tree species**[2,3] compared with clearcutting.

Background

Shelterwood harvesting is a management technique designed to obtain even-aged trees without clearcutting. It involves harvesting trees in a series of partial cuts, with trees removed uniformly over the plot. This allows new seedlings to grow from seeds dispersed by older trees. This can help in maintaining distinctive forest species and increase structural diversity of stands.

A replicated study in 1985-1993 in temperate coniferous forest in Sweden (1) found that **shelterwood harvesting increased plant diversity compared with clearcutting**. Plant diversity was higher in shelterwood (Simpson index: 0.48) than in clearcut areas (0.37). Species richness, average height and total cover of plants were similar between shelterwood (species: 5.3/0.25 m^2; height: 33 cm; cover: 75%) and clearcut areas (species: 4.2/0.25 m^2: height: 34 cm: cover: 65%). In 1985, 2-4 clearcut (all trees removed) plots (40 × 25 m) and 4-8 shelterwood (140-200 trees/ha retained) plots (20 × 25 m) were established in each of four sites. Monitoring was undertaken in 1993 in 24-160 subplots/treatment in each site. Each subplot was 0.5 × 0.5 m.

A replicated, controlled study in 1974-1992 in temperate coniferous forest in Montana, USA (2) found that **shelterwood harvesting increased the density of conifers compared with clearcutting**. Density (trees/ha) of conifers >30 cm tall (shelterwood: 19,895; clearcut: 6,834) and total conifer density (shelterwood: 31,389; clearcut: 8,741) were the highest in shelterwood. Three blocks of each treatment, shelterwood and clearcutting, were duplicated in two sites in 1974. Data were collected in 1992 in 80 plots (0.004 ha) in each treatment block.

A replicated, study in 1993-2000 in temperate forest in Sweden (3) found that **shelterwood harvesting increased the density of some tree species and decreased the cover of grasses compared with clearcutting**. Density (seedlings/ha) of Scots pine *Pinus sylvestris* (shelterwood: 18,500-23,000; clearcut: 3,000-6,500) and Norway spruce *Picea abies* (shelterwood: 17,000-20,000; clearcut: 2,500-3,000) was higher in shelterwood while density of birch Downy birch *Betula pubescens and Silver birch B. pendula* was similar between treatments (3,500-8,500 seedlings/ha). Cover of grasses (shelterwood: 19-20%; clearcut: 32-35%) was lower in shelterwood while cover of herbs (5-11%) and dwarf-shrubs (12-18%) was similar. In 1993-1995 two shelterwood (cutting 40% of volume) and two clearcut treatment plots (0.4 ha) were established in each of eight sites. Data were collected in 2000.

(1) Hannerz, M. and Hånell, B. (1997) Effects on the flora in Norway spruce forests following clearcutting and shelterwood cutting. *Forest Ecology and Management,* 90, 29-49.

(2) Shearer, R.C. and Schmidt, J.A. (1999) Natural regeneration after harvest and residue treatment in a mixed conifer forest of northwestern Montana. *Canadian Journal of Forest Research*, 29, 274-279.
(3) Karlsson, M., and Nilsson, U. (2005) The effects of scarification and shelterwood treatments on naturally regenerated seedlings in southern Sweden. *Forest Ecology and Management,* 205, 183-197.

5.5. Use partial retention harvesting instead of clearcutting

- Three studies (including one replicated, randomized, controlled study) in Canada[1,2,3] found that using partial retention harvesting instead of clearcutting decreased the **density of young trees**.

Background
Alternatives to traditional clearcut harvesting have been examined due to a recognized need to protect forests. Partial retention harvesting, i.e. retention of mature trees at harvest, has become commonly employed to maintain stand structural diversity and therefore sustain biodiversity.

A replicated study in 1999-2000 in boreal forest in Alberta, Canada (1) found that **partial retention harvesting decreased the cover and root-sucker density of *Populus spp.* compared with clearcutting**. Cover of *Populus spp.* (clearcutting: 9%; thinning: 3%) and density of *Populus spp.* root-suckers (stems/ha) (clearcutting: 74,800; thinning: 53,900) were higher in clearcut sites. Data were collected in August 2000 in twenty 2 x 2 m plots in each of three replicates of clearcutting (complete removal) and partial removal (50% of basal area removed) treatment units (10 ha). Treatments were applied during the winter of 1998-1999.

A replicated, controlled study in 1999-2007 in boreal mixed wood forest in Alberta, Canada (2) found that **low logging intensity levels decreased young tree density compared with clearcutting**. Young tree density for the dominant species trembling aspen *Populus tremuloides* and balsam poplar *Populus balsamifera* was higher in clearcutt plots (15,000/ha) than in 50% (7,000) and 75% tree area retention sites (3,000), and higher in 10% (12,000) than in 75% tree retention sites. In 20% retention sites, density was similar to all other retention levels (9,000). Each of five logging intensity levels: clearcutting (0%), 10%, 20%, 50% and 75% of the area retained were applied in nine 10 ha forest compartments. Regeneration of trembling aspen and balsam poplar was assessed using six 2 × 10 m plots in each compartment (total of 270 plots).

A replicated, randomized, controlled study in 1994-2005 in temperate coniferous woodland in British Columbia, Canada (3) found that **partial-cutting decreased the cover of understory conifers compared with clearcutting at one of two sites**. At one site cover of tall (2-10 m) understory conifers was higher in clearcut (12%) than partial-cut plots (3-6%). Cover of other plants was similar between treatments: tall shrubs (clearcut: 19%; partial-cut: 13-15%), short (<2 m) shrubs (clearcut: 39%; partial-cut: 24-28%), short conifers (clearcut: 2%; partial-cut: 4-5%), herbs (clearcut: 37%; partial-cut: 36-41%) and mosses and lichens (clearcut: 6%; partial-cut: 12-22%). At a second site, cover of plant groups was similar between treatments: tall conifers (clearcut: 8%; partial-cut: 6-8%), short conifers (clearcut: 10%; partial-cutt: 10-14%), tall shrubs (clearcut: 12%; partial-cut: 3-4%), short shrubs (clearcut: 41%; partial-cut: 19-28%), herbs (clearcut: 23%; partial-cut: 21-22%) and mosses and lichens (clearcut: 6%; partial-cut: 13-19%). Data were collected in 2004-2005 in 16 subplots (200 m^2) within each of four clearcut and eight partial cut (25-50% of basal area retained) treatment plots (1 ha) in each of two sites. Treatments were applied in 1994-1996.

(1) Frey, B.R., Lieffers, V.J., Munson, A.D., and Blenis, P.V. (2003) The influence of partial harvesting and forest floor disturbance on nutrient availability and understory vegetation in boreal mixedwoods. *Canadian Journal of Forest Research,* 33, 1180-1188.
(2) Gradowski, T., Lieffers, V.J., Landhäusser, S.M., Sidders, D., Volney, J., and Spence, J.R. (2010) Regeneration of Populus nine years after variable retention harvest in boreal mixedwood forests. *Forest Ecology and Management,* 259, 383-389.
(3) Newsome, T.A., Heineman, J.L., Nemec, A.F.L., Comeau, P.G., Arsenault, A., and Waterhouse, M. (2010) Ten-year regeneration responses to varying levels of overstory retention in two productive southern British Columbia ecosystems. *Forest Ecology and Management,* 260, 132-145.

5.6. Use summer instead of winter harvesting

- One replicated study in the USA[1] found no effect of logging season on **plant species richness and diversity**.

Background
Logging alters the composition and ecology of forests. In seasonal climates the timing of logging operations can be adjusted to avoid damage during the major growing season of understory plants and seedlings.

A replicated study in 2002-2006 in temperate mixed forest in Wisconsin, USA (1) found **no effect of logging season on plant species richness and diversity**. Plant species richness/plot (winter-logged: 71-103; summer-logged: 70-99) and Shannon's index of diversity (winter-logged: 5.5-6.0; summer-logged: 5.5-6.0) were similar between treatments. Five plots (100 × 250 m) were logged in 2002-2005 to retained 16-19 m^2/ha basal area in summer (June - October) and winter (November – March). Data were collected in May-September 2006 in 60 plots (2 × 2 m) in each treatment plot.

(1) Wolf, A.T., Parker, L., Fewless, G., Corio, K., Sundance, J., Howe, R., and Gentry, H. (2008) Impacts of summer versus winter logging on understory vegetation in the Chequamegon-Nicolet National Forest. *Forest Ecology and Management,* 18, 414-417.

5.7. Adopt continuous cover forestry

- We found no evidence for the effects of adopting continuous cover forestry on forests.

Background
Continuous cover forestry is a way of managing forests where clearcutting is avoided in favour of other management systems that maintain a continuous

forest cover and high structural diversity. This may increase species diversity compared to clearcutting.

5.8. Use brash mats during harvesting to avoid soil compaction

- We found no evidence for the effects of using brash mats during harvesting to avoid soil compaction on forests.

Background

Using heavy machinery during harvesting may compact the soil and limit seedling regeneration. Remaining woody debris from harvested trees (brash) can be used to cover the ground and form mats. This may spread out the weight of heavy machinery and decrease soil compaction.

Harvest forest products

5.9. Sustainable management of non-timber forest products

- We found no evidence for the effects of sustainable management of non-timber forest products on forests.

5.10. Adopt certification

- One replicated, site comparison study in Ethiopia[2] found that the risk of deforestation was lower in certified than uncertified forests. One controlled, before-and-after trial in Gabon[1] found that when logging intensity was taken into account although tree damage did not differ, changes in above-ground biomass were smaller in certified than in uncertified forests.

Background

Forest certification is a market-based mechanism that tries to ensure sustainable wood harvesting. Well-known examples are the *Forest Stewardship Council* (FSC) or *Programme for the Endorsement of Forest Certification* (PEFC). To receive certification, forests need to be managed according to a pre-agreed set of standards to ensure their sustainability (standards may vary depending on the type of certification). Certification allows foresters to identify themselves as sustainable producers and add a price premium to their products. This could make their business more profitable possibly encouraging other foresters to adopt certification and potentially benefiting biodiversity.

A controlled, before-and-after study in 2010–2011 in two mixed lowland tropical forests in Ogooué-Ivindo, Gabon (1) found that **once logging intensity was taken into account a certified logged forest had similar tree damage but a smaller change in above-ground biomass than a more intensively logged**

uncertified forest. The amount of tree biomass damaged was lower in the certified forest than in the uncertified forest (certified: 18.7; uncertified: 33.7 Mg/ha). However, there was no difference in damage between the two forests when the higher logging intensity in the conventional forest was taken into account (certified: 3.3; uncertified: 2.9 Mg/m^3). The change in above-ground biomass was smaller in the certified forest (certified: 7%; uncertified: 13%), even when corrected for logging intensity. The tree species composition after logging did not change in either forest (difference in Simpson's Index before and after harvest: certified 0.96; uncertified 0.41). The logging intensity in the certified forest was 5.7 m^3/ha and 11.4 m^3/ha in the uncertified forest. Twenty plots in the certified forest and 12 in the uncertified forest were established (each 200 × 50 m). Measurements were taken within each plot 2-6 months before and 2-3 months after logging for all trees with a diameter breast height > 10 cm.

A replicated, site comparison study in 2010 in highland rainforests in the Oromiya region, Ethiopia (2) found that **forests producing wild, shade-grown coffee *Coffea arabica* with a certification had a lower risk of deforestation than forests where coffee was grown without certification**. Forests under a coffee certification program had a lower probability of deforestation (2.8%) than similar areas where no forest coffee was produced (4.5%). However, where coffee was grown without certification, the probability of deforestation (11.8%) did not differ from similar areas where no coffee was grown (12.4%). The study was conducted in two forests that were certified in 2007 and two forests that were considered uncertified during the study as they only received certification in 2009, just before the measurements of forest cover in 2010. Probability of deforestation was estimated using satellite images (Landsat, resolution 30 m) from 2005 and 2010.

(1) Medjibe V. P., Putz F. E. & Romero C. (2013). Certified and uncertified logging concessions compared in Gabon: changes in stand structure, tree species, and biomass. *Environmental Management,* 51, 524–540.
(2) Takahashi R. & Todo Y. (2013) The impact of a shade coffee certification program on forest conservation: A case study from a wild coffee forest in Ethiopia. *Journal of Environmental Management,* 130, 48–54.

Firewood

5.11. Provide fuel efficient stoves

- We found no evidence for the effects of providing fuel efficient stoves on forests.

Background
In many areas wood is the main fuel used for cooking and heating. The high demand for firewood can lead to strong pressure on forests. By providing fuel-efficient stoves, less wood is required, which may lead to reduced exploitation of forests.

5.12. Provide paraffin stoves

- We found no evidence for the effects of providing paraffin stoves on forests.

Background

In many areas wood is the main fuel used for cooking and heating. The high demand for firewood can lead to strong pressure on forests. By providing alternative stoves that do not require wood (such as paraffin stoves), less wood is required, which may lead to reduced exploitation of forests.

6. Natural system modification

Natural forest systems are affected by human activities and modified in many ways. In forested areas, trees were used as traditional materials for constructions, preparing tools and fuel. This utilization of trees varied from total clear cutting to light selective wood harvesting. Since man acquired control over fire, fires had tremendous effect on forests and still have. Even in places where wildfires occur naturally as result of lightning, arson or neglect, fires have greater effect on forests. However, prescribed fires are often used as a tool for reducing plant biomass and reducing the risk of future fires.

Key messages - changing fire frequency

Use prescribed fire

Four of nine studies, including two replicated, randomized, controlled studies, in the USA found that prescribed fire decreased mature tree cover, density or diversity. Two studies found it increased tree cover or size, and four found mixed or no effect. Seven studies, including one replicated, randomized, controlled study, in the USA found that fire increased mature tree mortality. Five of 15 studies, including four replicated, randomized, controlled studies, in France, Canada and the USA found that prescribed fire increased the density and biomass of young trees. Two found that fire decreased young tree density. Eight found mixed or no effect on density and two found mixed effects on species diversity of young trees. Two replicated, controlled studies in the USA found mixed effects of prescribed fire on young tree survival. Eight of 22 studies, including seven replicated, randomized, controlled studies, in Australia, Canada and the USA found that prescribed fire increased the cover, density or biomass of understory plants. Six found it decreased plant cover and eight found mixed or no effect on cover or density. Fourteen of 24 studies, including 10 replicated, randomized, controlled studies, in Australia, France, West Africa and the USA found that fire increased species richness and diversity of understory plants. One found it decreased species richness and nine found mixed or no effect on understory plants.

Use herbicides to remove understory vegetation to reduce wildfires

We found no evidence for the effects of using herbicides to remove understory vegetation to reduce wildfires on forests.

Mechanically remove understory vegetation to reduce wildfires

We found no evidence for the effects of using herbicides to remove understory vegetation to reduce wildfires on forests.

Key messages – water management

Recharge groundwater to restore wetland forest

We found no evidence for the effects of recharging groundwater to restore wetland forest on forests.

Construct water detention areas to slow water flow and restore riparian forests
We found no evidence for the effects of constructing water detention areas to slow water flow and restore riparian forests on forests.

Introduce beavers to impede water flow in forest watercourses
We found no evidence for the effects of introducing beavers to impede water flow in forest watercourses on forests.

Key messages - changing disturbance regime

Use clearcutting to increase understory diversity
Three of nine studies, including four replicated, randomized, controlled studies, in Australia, Japan, Brazil, Canada and the USA found that clearcutting decreased density, species richness or diversity of mature trees. One study found it increased trees species richness and six found mixed or no effect or mixed effect on density, size, species richness or diversity. One replicated, randomized, controlled study in Finland found that clearcutting decreased total forest biomass, particularly of evergreen shrubs. Three of six studies, including five replicated, randomized, controlled studies, in Brazil, Canada and Spain found that clearcutting increased the density and species richness of young trees. One found it decreased young tree density and two found mixed or no effect. Eight of 12 studies, including three replicated, randomized, controlled studies, across the world found that clearcutting increased the cover or species richness of understory plants. Two found it decreased density or species richness, and two found mixed or no effect.

Use shelterwood harvesting
Six of seven studies, including five replicated, controlled studies, in Australia, Iran, Nepal and the USA found that shelterwood harvesting increased abundance, species richness or diversity or understory plants, as well as the growth and survival rate of young trees. One study found shelterwood harvesting decreased plant species richness and abundance and one found no effect on abundance. One replicated, controlled study in Canada found no effect on oak acorn production.

Use group-selection harvesting
Four of eight studies, including one replicated, controlled study, in Australia, Canada, Costa Rica and the USA found that group-selection harvesting increased cover or diversity of understory plants, or the density of young trees. Two studies found it decreased understory species richness or and biomass. Three studies found no effect on understory species richness or diversity or tree density or growth-rate.

Use herbicides to thin trees
One replicated, controlled study in Canada found no effect of using herbicide to thin trees on total plant species richness.

Thin trees by girdling (cutting rings around tree trunks)
One before-and-after study in Canada found that thinning trees by girdling increased understory plant species richness, diversity and cover.

Use thinning followed by prescribed fire
Three of six studies,including one replicated, randomized, controlled study, in the USA found that thinning followed by prescribed fire increased cover or abundance of understory plants, and density of deciduous trees. One study found it decreased tree density and species richness. Three studies found mixed or no effect or mixed effect on tree growth rate or density of young trees. One replicated, controlled study Australia found no effect of thinning then burning on the genetic diversity of black ash.

Reintroduce large herbivores
We found no evidence for the effects of reintroducing large herbivores on forests.

Pollard trees (top cutting or top pruning)
We found no evidence for the effects of pollarding trees on forests.

Coppice trees
We found no evidence for the effects of coppicing trees on forests.

Halo ancient trees
We found no evidence for the effects of haloing ancient trees on forests.

Adopt conservation grazing of woodland
We found no evidence for the effects of adopting conservation grazing of woodland on forests.

Retain fallen trees
We found no evidence for the effects of retaining fallen trees on forests.

Imitate natural disturbances by pushing over trees
We found no evidence for the effects of imitating natural disturbances by pushing over trees on forests.

Changing fire frequency

6.1. Use prescribed fire

Background
Prescribed fires are undertaken to reduce the amount of combustible fuel in an attempt to reduce the risk of more extensive, potentially more damaging 'wildfires'. They may also be used for maintenance or restoration of habitats historically subject to occasional 'wildfires' that have been suppressed through management. Prescribed fires may remove large amounts of woody material from the forest understory and increase the amount of grasses and other herbaceous vegetation.

6.1.1. Use prescribed fire: effects on mature trees

- Four of eight studies (including two replicated, randomized, controlled studies) from the USA found that prescribed fire decreased **tree cover**[4], **density**[2,5,14] and **diversity**[2]. One study[6] found it increased tree cover and three[9,10,13] found no effect or mixed effects of prescribed fire on cover and density of trees.
- Seven studies from the USA[3,7,8,11,12,15,16] (including one replicated, randomized, controlled study) found that prescribed fire increased **tree mortality**.
- One of three studies from the USA (including one replicated, controlled study) found that prescribed fire increased **tree size**[10] while two[1,9] found no effect of prescribed fire on tree size.

A controlled study in 1991-1997 in temperate coniferous forest in Louisiana, USA (1) found **no effect of prescribed burning on longleaf pine *Pinus palustris* growth and yield**. Average diameter at breast height (30-31 cm), total height (23 m in both) and basal area (23-24 m^2/ha) of longleaf pine were similar between treatments. Data were collected in four replicates of 0.16 ha burned (prescribed burned in March 1991, February 1994 and March 1997) and unburned control treatment plots. Longleaf pine were sampled in February 1996 in one 0.09 ha subplot within each plot.

A before-and-after trial in 1994-1995 in temperate mixed forest in North Carolina, USA (2) found that **prescribed fire decreased the density and diversity of canopy trees** in one of three locations. In one site located on top of the ridge density of trees >5 cm diameter at breast height decreased after burning (before: 1,545; after: 913/ha) as well as their diversity (Shannon's index before: 1.92; after: 1.73). In two other sites located in the middle of the slope and next to the stream tree density remained similar (before: 1,167-1,448; after: 1,117-1,365/ha) as well as their diversity (Shannon's index before: 1.85-2.25; after: 1.81-2.26). Data were collected before (1994) and after prescribed-burning (1995) in six plots (15 × 15 m) at one location (top of the ridge) and in three similar plots at two other locations (middle of the slope and close to the stream) and in additional 20 plots (10 × 10 m) after burning.

A replicated, controlled study in 1984-1995 in temperate broadleaf woodland in Minnesota, USA (3) found that **prescribed fires increased mortality of northern pin oak *Quercus ellipsoidalis* but not of bur oak *Q. macrocarpa***. Mortality of northern pin oak was higher in burned (50%, 560 trees) than unburned plots (27%, 293 trees), while mortality rate of bur oak was similar in burned (8%, 120 trees) and unburned plots (17%, 40 trees). Data were collected in 1995 in 11 burned (4-26 prescribed fires, 1984-1995) and eight unburned plots (0.38 ha).

A replicated, randomized, controlled study in 1997-2001 in chaparral in Texas, USA (4) found that **prescribed fire decreased woody plant cover and species diversity but not species richness**. Woody plant cover was higher in unburned (44%) than winter (22%) and winter summer burn plots (26%). Diversity was higher in unburned (Shannon's index unburned: 2.55) than winter and summer burn plots (winter burn: 2.33; winter and summer burn: 2.13).

Numbers of species was similar among treatments (unburned: 18; winter burn: 16; winter and summer burn: 13/2 ha plot). Data were collected in 2001 in five replicates of: unburned control, winter burn (winter 1997-1998 and 1999-2000) and winter and summer burn (burned in winter 1997-1998 and summer 1999) treatment plots (2 ha).

A replicated, controlled study in 2000-2005 in temperate coniferous forest in California, USA (5) found that **prescribed fire decreased tree density, but not basal area, height and canopy cover**. The density for all trees >2.5 cm diameter at breast height was lower in burned (441 trees/ha) than unburned plots (1,109/ha), while basal area (burned: 48; unburned: 56 m^2/ha), height (burned: 18; unburned: 16 m) and canopy cover (burned: 65; unburned: 75%) were similar between treatments. Data were collected in 2005 in 25 plots (0.04 ha) in each of three burned (prescribed fire in October-November 2002) and three unburned control treatment unit (14-29 ha).

A replicated, controlled study in 2001 in temperate broadleaf forest in Tennessee, USA (6) found that **prescribed fire increased tree canopy cover but not species diversity**. Canopy cover was higher in burned plots (burned: 4%; unburned: <1%), while diversity (Simpson's index) of herbs (burned: 2.6; unburned: 3.1) and woody plants (burned: 3.1-3.3; unburned: 2.3-2.8) was similar between treatments. Data were collected in summer 2001 in two burned (prescribed fire in April 2001) and two control (unburned) treatment plots (0.8 ha) in each of four sites.

A replicated, randomized, controlled study in 2001-2004 in temperate coniferous forest in Florida, USA (7) found that **prescribed fire increased longleaf pine *Pinus palustris* mortality when the leaf litter layer was dry** on the day of burn, **but not when it was moist**. Longleaf pine mortality was higher following burns with dry leaf litter than following the other three treatments (dry litter: 16%; moist litter: 4%; wet litter: 2%; unburned: 0%). In 2001-2002, four treatment plots (10-50 ha) were established at each of four sites: dry, moist and wet leaf litter (55%, 85% and 115% litter moisture content, % of dry mass, on the day of burn) and an unburned control. Data were collected two years after treatments.

A paired-sites study in 2003-2006 in temperate coniferous forest in Arizona and New-Mexico, USA (8) found that **prescribed burns increased tree mortality**. Mortality of ponderosa pine *Pinus ponderosa* increased from 0.6% in unburned to 8.4% in burned plots and mortality of all tree species increased from 0.6% in unburned to 9.6% in burned forest units by the end of the third growing season after burning. Trees with at least a 13 cm diameter at breast height were monitored for three growing seasons after burning (2004–2006). They were monitored in 25-40 circular plots (10 m radius within each burned (prescribed fire between 2003 and 2004) and unburned control forest units in each of four sites.

A replicated, controlled study in 2002-2006 in temperate coniferous forest in Washington State, USA (9) **found no effect of prescribed fire on tree density, basal area, average diameter and height**. Numbers of trees (burned: 525; unburned: 530/ha), tree basal area (burned: 34; unburned: 34 m^2/ha), average diameter (burned: 31; unburned: 30 cm) and height (burned: 6; unburned: 5 m) were similar between treatments. Data were collected in 2006 in six plots (20 ×

50 m) within each of six burned (prescribed fire in spring 2004) and six unburned treatment units (10 ha).

A before-and-after trial in 2003-2006 in temperate coniferous forest in California, USA (10) found that prescribed fire **increased tree height and diameter but did not affect their density and cover**. Tree height at the base of the canopy (before: 3.7-4.3 m; after: 6.7-7.3 m) and average diameter (before: 36 cm; after: 38 cm) increased after treatments. However, the number of trees (before: 408-462; after: 351-356/ha) and canopy cover (before: 49-74%; after: 46-74%) remained similar. Data were collected in 42 plots (0.1 ha) before (2003) and after (2006) the prescribed fire in 2003.

A randomized, controlled study in 2004-2008 in temperate coniferous forest in California USA (11) found that **prescribed burning increased tree mortality**. The overall percentage of trees killed was higher following early and late-season burning (16% and 18%) than in unburned plots (1%). For trees between 10-20 and between 31-41 cm diameter at breast height, mortality was higher following late-season burning (65% and 5% respectively) than in unburned plots (1% in both categories) and similar to both following early-season burning (22% and 4% respectively). For trees >51 cm diameter at breast height mortality was higher following early-season burning (4%) than in unburned plots (0%) and similar to both following late-season burning (2%). For trees 21-30 and 41-51 cm diameter at breast height mortality was similar in unburned (1% and 0% respectively), early-season (11% and 3% respectively) and late-season burning plots (8% and 2% respectively). Three treatments: control (unburned); early-season prescribed burn (May 2005); late-season prescribed burn (October 2005) were each randomly assigned to three 4 ha plots. Tree mortality was monitored in 2005-2008.

A replicated, controlled study in 2002-2006 in temperate coniferous forest in California, USA (12) found that p**rescribed burning increased bark-beetle caused tree mortality**. Cumulative percentage of trees killed by bark beetles were higher in burned (9%) than in unburned plots (3%). All trees killed by bark beetles were recorded in 2006 in three control (unburned) and three burned (prescribed burning in the autumn) treatment units (10 ha). Prescribed burning was in 2002.

A replicated, controlled, randomized study in 2001-2005 in temperate coniferous forest in Montana, USA (13) **found no effect of prescribed fire on trees density and basal area**. Density of trees (burned: 386; control: 400/ha) and tree basal area (burned: 22; control: 25 m^2/ha) were similar between treatments. Data were collected in 2005 in ten 0.1 ha plots in each of three replicates of burned (prescribed broadcast burning in spring 2002) and control (unburned) treatments.

A replicated, before-and-after study in 2006-2010 in temperate coniferous forest in Washington State, USA (14) found that **prescribed burning decreased the density, but not basal area of trees**. The density of all trees (pre-fire: 1,565; post-fire: 655/ha) and of trees <20 cm diameter at breast height (pre-fire: 845; post-fire: 190/ha) was lower three years post-fire. Basal area of trees was similar (pre-fire: 82; post-fire: 60 m^2/ha). Data were collected in 2006 (pre-fire) and in 2010 (post-fire) in five 20 x 20 m plots that were burned in 2007.

A replicated, controlled study in 2001-2009 in temperate coniferous forest in California, USA (15) found that **prescribed burning increased tree mortality**

in the short-term but not in the long-term. Tree mortality rate was higher in burned plots in the first six years after treatment (unburned: 1.5% in all six years; burned: 47% in the first to 3% in the sixth year), but similar in the following two years (unburned: 1.5%; burned: 1.5-2.0%). Data were collected in 2001-2009 in five burned (prescribed-fire in 2001) and seven unburned control plots (0.9-2.5 ha).

A replicated, controlled study in 2001-2003 in temperate coniferous forest in California, USA (16) found that **prescribed burning increased bark beetle caused mortality of white fir *Abies concolor* trees but not of sugar pine *Pinus lambertiana* or ponderosa pine *Pinus ponderosa***. Mortality of white fir trees 11-25 cm diameter at breast height (burned: 4.6%; unburned: 0.2%) and trees 25-45 cm diameter at breast height (burned: 0.8%; unburned: 0.1%) was higher in burned than unburned plots. Mortality did not differ between treatments for sugar pine 11-25 cm diameter at breast height (burned: 2.9%; unburned: 0.0%) and 25-45 cm diameter at breast height (burned: 4.8%; unburned: 0.0%), and of ponderosa pine 11-25 cm diameter at breast height (burned: 1.8%; unburned: 0.0%) and 25-45 cm diameter at breast height (burned: 0.0%; unburned: 0.0%). For trees >45 cm diameter at breast height, mortality did not differ between treatments for either white fir (burned: 0.3%; unburned: <0.1%), sugar pine (burned: 0.0%; unburned: 0.0%) or ponderosa pine (burned: 0.1%; unburned: 0.0%). Bark beetle caused tree mortality was monitored in 2003 in 20 subplots (0.4 ha) in each of three unburned control and three burned (prescribed fire in November 2002) treatment plots (14-29 ha).

(1) Haywood, J.D., Tiarks, A.E., Elliott-Smith, M.L., and Pearson, H.A. (1998) Response of direct seeded Pinus palustris and herbaceous vegetation to fertilization, burning, and pine straw harvesting. *Biomass and Bioenergy,* 14, 157-167.

(2) Elliott, K.J., Hendrick, R.L., Major, A.E., Vose, J.M., and Swank, W.T. (1999) Vegetation dynamics after a prescribed fire in the southern Appalachians. *Forest Ecology and Management*, 114, 199-213.

(3) Peterson, D.W., and Reich, P.B. (2001) Prescribed fire in oak savanna: fire frequency effects on stand structure and dynamics. *Ecological Applications,* 11, 914-927.

(4) Ruthven III, D.C., Braden, A.W., Knutson, H.J., Gallagher, J.F., and Synatzske, D.R. (2003) Woody vegetation response to various burning regimes in South Texas. *Journal of Range Management,* 56, 159-166.

(5) Stephens, S.L., and Moghaddas, J.J. (2005) Experimental fuel treatment impacts on forest structure, potential fire behavior, and predicted tree mortality in a California mixed conifer forest. *Forest Ecology and Management,* 215, 21-36.

(6) Jackson, S.W., Harper, C.A., Buckley, D.S., and Miller, B.F. (2006) Short-term effects of silvicultural treatments on microsite heterogeneity and plant diversity in mature Tennessee oak-hickory forests. *Northern Journal of Applied Forestry,* 23, 197-203.

(7) Varner III, J.M., Hiers, J.K., Ottmar, R.D., Gordon, D.R., Putz, F.E., and Wade, D.D. (2007) Overstory tree mortality resulting from reintroducing fire to long-unburned longleaf pine forests: The importance of duff moisture. *Canadian Journal of Forest Research,* 37, 1349-1358.

(8) Breece, C.R., Kolb, T.E., Dickson, B.G., McMillin, J.D., and Clancy, K.M. (2008) Prescribed fire effects on bark beetle activity and tree mortality in southwestern ponderosa pine forests. *Forest Ecology and Management,* 255, 119-128.

(9) Harrod, R.J., Peterson, D.W., Povak, N.A. , and Dodson, E.K. (2009) Thinning and prescribed fire effects on overstory tree and snag structure in dry coniferous forests of the interior Pacific Northwest. *Forest Ecology and Management,* 258, 712-721.

(10) Vaillant, N.M., Fites-Kaufman, J., Reiner, A.L., Noonan-Wright, E.K., and Dailey, S.N. (2009) Effect of fuel treatments on fuels and potential fire behavior in California, USA national forests, *Fire Ecology*, 5, 14-29.

(11) Fettig, C.J., McKelvey, S.R., Cluck, D.R., Smith, S L., and Otrosina, W.J. (2010) Effects of prescribed fire and season of burn on direct and indirect levels of tree mortality in Ponderosa and Jeffrey Pine Forests in California, USA. *Forest Ecology and Management,* 260, 207-218.
(12) Fettig, C., Borys, R., and Dabney, C. (2010) Effects of fire and fire surrogate treatments on bark beetle-caused tree mortality in the Southern Cascades. California, *Forest Science,* 56, 60-73.
(13) Fiedler, C.E., Metlen, K.L., and Dodson, E.K. (2010) Restoration treatment effects on stand structure, tree growth, and fire hazard in a ponderosa pine/douglas-fir forest in Montana. *Forest Science,* 56, 18-31.
(14) Fonda, R.W., and Binney, E.P. (2011) Vegetation response to prescribed fire in Douglas-fir forests, Olympic National Park. *Northwest Science,* 85, 30-40.
(15) Van Mantgem, P.J., Stephenson, N.L., Knapp, E., Battles, J., and Keeley, J.E. (2011) Long-term effects of prescribed fire on mixed conifer forest structure in the Sierra Nevada. California, *Forest Ecology and Management,* 261, 989-994.
(16) Stark, D.T., Wood, D.L., Storer, A.J., and Stephens, S.L. (2013) Prescribed fire and mechanical thinning effects on bark beetle caused tree mortality in a mid-elevation Sierran mixed-conifer forest. *Forest Ecology and Management*, 306, 61-67.

6.1.2. Use prescribed fire: effects on young trees

- Five of 15 studies (including four replicated, randomized, controlled studies) from Canada[16], France[12] and the USA[1,2,3,4,5,6,7,9,10,11,13,14,17], found that prescribed fire increased the **density and biomass of young trees**[3,4,6,12,16]. Two studies found that fire decreased new tree density[1,14]. Eight found no effect[9,10,17] or mixed effects depending on the tree species[7,11], location[2,13] and fire frequency[5].
- Two of the above studies found mixed effects of prescribed fire on **species diversity of young trees** depending on the location[2,13].
- Two replicated, controlled studies from the USA[8,15] found mixed effects of prescribed fire on the **survival of young trees**.

A replicated, randomized, controlled study in 1995-1996 in temperate broadleaf forest in Virginia USA (1) found that **prescribed burning reduced densities of new trees of all hardwood species**. Declines in density were greater following summer burns than spring or winter burns and smallest in unburned plots for hickory *Carya spp.* (summer burn: 1,105; spring burn: 662; winter burn: 643; unburned: 76 trees/ha) and oak *Quercus spp.* (summer burn: 1,124; spring burn: 543; winter burn: 531; unburned: 79). Declines in density were higher in summer and spring burn than in winter burn and the lowest in control of red maple *Acer rubrum* (summer burn: 1,475; spring burn: 1,425; winter burn: 541; unburned: 82/ha) and yellow-poplar *Liriodendron tulipifera* (summer burn: 4,231; spring burn: 4,169; winter burn: 2,801; unburned: 70) were. In 1995, four 2-5 ha areas were randomly assigned to one of four burn treatments: winter (February), spring (April) and summer (August) fires and unburned, in each of three forest sections. Each section had been shelterwood harvested. Monitoring was carried out in the autumn of 1994 (before treatments), 1995 and 1996 (after treatments) in 15 plots (20 m^2) in each treatment.

A before-and-after trial in 1994-1995 in temperate mixed forest in North Carolina, USA (2) found that **prescribed fire had mixed effects on the density and diversity of young trees** in three study sites. In a site located on top of the ridge density of young trees <5 cm diameter at breast height decreased after burning (before: 12,178; after: 409/ha) while their diversity remained similar

(Shannon's index before: 1.24; after: 1.27. In a second site located in middle of the slope density of young trees increased (before: 851; after: 1,556/ha) while diversity decreased (Shannon's index before: 1.52; after: 0.54). In a third site located close to the stream density (before: 2,153; after: 2,652/ha) and diversity (Shannon's index before: 2.15; after: 2.40) remained similar after burning. Data were collected before (1994) and after prescribed-burning (1995) in six plots (15 × 15 m) at one location (top of the ridge) and in three similar plots at two other locations (middle of the slope and close to the stream) and in additional 20 plots (10 × 10 m) after burning.

A replicated, controlled study in 1995-1998 in temperate mixed forest in Florida, USA (3) found that **prescribed burning increased the density of oak *Quercus spp.* but not of longleaf pine *Pinus palustris* juveniles**. Numbers of oak juveniles was higher in burned (2.0/m^2) than unburned plots (1.5/m^2), while numbers of longleaf pine juveniles was similar in burned (90/200 m^2) and unburned plots (75/200 m^2). One burned (prescribed burned in spring 1995) and one unburned control plots (81 ha) were established in 1995 in each of six blocks. Data were collected in 1998 in 32 subplots (40 × 10 m) in each plot.

A replicated, controlled study in 2001-2003 in temperate mixed forest in Georgia and Tennessee, USA (4) found that **prescribed burning increased the density of trees <5 cm diameter at breast height but not of larger trees and did not affect species richness of new trees**. Density of trees <5 cm diameter at breast height was higher in burned sites (burned: 122,660; unburned: 63,560 stems/ha). However, density of trees >5 cm diameter at breast height was similar in burned (1,150) and unburned sites (870). The number of species of trees did not differ between treatments for those <5 cm diameter at breast height (burned: 29.8; unburned: 27.0) and those >5 cm diameter at breast height (burned: 15.8; unburned: 15.0). Data were collected in 2002 in five 10 × 20 m plots in each of four burned (prescribed burned in March 2001) and two control (unburned) sites (total of 30 plots).

A replicated, controlled study in 1994-2002 in temperate oak forest in Ohio, USA (5) found that **two consecutive annual burnings decreased the number of small trees compared with no burning or four annual burnings; both two and four annual burnings decreased the density of saplings**. The density of small trees was lower in burned×2 (130 individuals/ha) than unburned (190) and burned×4 plots (170). The density of large and small saplings was higher in unburned (600 and 1,100 respectively) than in burned×4 plots (200 and 100-250 respectively). Three treatment units (25 ha) were replicated in four sites: burned×4 (burned annually 1996-1999), burned×2 (burned in 1996 and 1999) and unburned. Data were collected in 2002 in nine plots (0.125 ha) within each treatment unit.

A replicated, controlled study in 1999-2003 in temperate broadleaf forest in South Carolina, USA (6) found that **prescribed fire increased the biomass and density of white oak *Quercus alba* seedlings**. Numbers of seedlings (burned: 0.03-1.5; unburned: 0.02-0.15/ha) and average seedling biomass (burned: 0.68 g; unburned: 0.43 g) were higher in burned plots. Three forest units were divided into burned (prescribed fires in 1999 and 2000) and control (unburned) treatments (>1 ha). Data were collected in 2003 under six to eight white oak trees in each forest unit.

A replicated, controlled study in 2001-2005 in second-growth oak forests in southern Ohio, USA (7) found that **prescribed fire reduced total large sapling density and increased large seedlings density, but not seedlings of oaks** *Quercus* spp. A single prescribed fire reduced large sapling (3-10 cm diameter at breast height) density from 600 to 300 saplings/ha and increased large seedlings (40-150 cm tall) density from 2,000 to 6,000 seedlings/ha. Prescribed fire had no effect on the densities of small seedlings <50 cm tall (control: 135,000; fire: 140,000 seedlings/ha) and small saplings <3 cm diameter at breast height (control: 1,000; fire: 1,050 saplings/ha). A single prescribed fire did not affect densities of oak seedlings. Three forest areas were divided into treatment units (each approximately 30 ha): control and prescribed fire. Treatments were applied in the inactive season of 2001. New tree growth was sampled in ten 0.1 ha plots/treatment (a total of 40 plots/site) in summer 2004.

A replicated, controlled study in 2002-2007 in temperate oak forest in Kentucky, USA (8) found that **prescribed fire and fire frequency had mixed effects on seedling survival depending on tree species**. White oak *Leucobalanus spp.* seedling survival decreased to 50% in single-burn and to 40% in repeated-burn plots, compared with 70% in unburned plots. Red oaks *Erythrobalanus spp.* seedling survival decreased to 65% in repeated-burn compared with 75% in unburned plots. Red maple *Acer rubrum* seedling survival decreased to 40% in single-burn and in repeated-burn plots compared with 80% in unburned plots. Sassafras *Sassafras albidum* seedling survival was not affected by burning. Three study sites (200-300 ha) were subdivided into three burn treatments (58–116 ha): an unburned control, single burn (spring 2003) and repeated burning (spring 2003, 2004 and 2006). Approximately 3,000 seedlings were tagged in June 2002 and survival monitored annually from 2002 to 2007 in 8–12 plots (10 x 40 m) within each treatment.

A site comparison study in 2002-2003 in temperate coniferous forest in Nevada USA (9) found **no effect of burning on the number of emerged seedlings**. Numbers of seedlings was similar in burned (743 seedlings/m^2) and unburned plots (635 seedlings/m^2). .Three 15 × 25 m plots were located in each of adjacent burned (prescribed fire in May 2002) and unburned areas. Seeds were sampled in thirty 0.1 × 0.1 × 0.05 m soil samples in each plot. Sampling was one week before burning and two growing seasons after.

A replicated, controlled study in 2000-2005 in temperate broadleaf forest in Ohio, USA (10) found **no effect of prescribed fire on the number of chestnut oak *Quercus prinus* and black oak *Quercus velutina* acorns**. Density was similar between treatments for both chestnut oak (burned: 300,000-350,000; unburned: 250,000-300,000 acorns/ha) and black oak (100,000-400,000 in both). Data were collected in 2005 in nine burned (prescribed fire in 2001 and 2005) and nine unburned plots (0.1 ha) in at each of two forest sites (40 ha).

A replicated, randomized, controlled study in 2000-2005 in temperate forest in California, USA (11) found that **prescribed fire increased conifer seedling and decreased oak seedling densities**. Combined conifer and California black oak *Quercus kelloggii* seedling density was higher in burned (14.0/m^2) than unburned plots (1.5/m^2) while density of California black oak was higher in unburned *plots* (burned: 0.10; unburned: 0.45/m^2). Data were collected in 2006 in 10 sets of four plots (1 m^2) in each of three unburned control and three burned (October-November 2002) treatment units (14-29 ha).

A replicated, randomized, controlled study in 2004-2008 in Mediterranean Aleppo pine *Pinus halepensis* woodland in France (12) found that **prescribed burning increased Aleppo pine seedling density where woody debris was left but not where it was removed**. Density of seedlings in plots with woody debris was higher when burned (2.1 seedlings/m^2) than unburned plots (<0.1). Where woody debris had been removed, density was similar between treatments (unburned: 0.1; burned: 0.4). Data were collected in January 2008 in 16 plots (14 × 14 m): eight control plots with no fire and eight that had a prescribed fire in 2005. Four control and four burn plots had woody debris and four of each treatment plots had the woody debris manually removed. All plots were thinned in 2004 (from 410 to 210 trees/ha).

A controlled study in 2004-2006 in temperate mixed forest in North Carolina and Georgia, USA (13) found that **prescribed fire had mixed effects on density and diversity of young trees depending on site**. At one site density of young trees <5 cm diameter at breast height was lower in burned plots (burned: 7,000; unburned: 31,000/ha), while their diversity was similar between treatments (Shannon's index burned: 0.90; unburned: 1.00). At a second site density of young trees was similar between treatments (burned: 9,000; unburned: 14,000) while their diversity was lower in burned plots (burned: 0.87; unburned: 1.44). At a third site, the density of young trees (burned: 3,500; unburned: 1,200) and their diversity (burned: 0.57; unburned: 0.27) were similar between treatments. Data were collected in 2006 in 10-12 plots (10 × 20 m) in a burned area (in 2004) and in 4-6 plots in an adjacent unburned area (control) in each of three sites.

A replicated, randomized, controlled study in 2001-2005 in temperate coniferous forest in Montana, USA (14) found that **prescribed fire decreased tree saplings density**. The density of saplings between 0.1 to 10 cm diameter at breast height was lower in burned plots (burned: 6,550; control: 11,483/ha). Data were collected in 2003 in ten 0.1 ha plots in each of three replicates of burned (prescribed broadcast burning in spring 2002) and unburned treatments.

A replicated, controlled study in 2002-2008 in temperate broadleaf forest in Kentucky, USA (15) found that **frequent prescribed fires decreased and single fires increased the height of white oak *Quercus alba* seedlings and decreased their mortality rate**. Fire did not affect chestnut oak *Quercus prinus* seedling size and mortality rate. The height of white oak seedlings was different between treatments (unburned: 12; single fire: 16; three fires: 10 cm), while their diameters were higher in single than three fire plots (unburned: 2.3; single fire: 1.8; three fires: 1.7 mm). Cumulative percent mortality for white oak seedlings was lower in single fire (65%) than unburned plots (85%). For chestnut oak seedling height (unburned: 15; single fire: 16; three fires: 13 cm) and diameter (unburned: 2.3; single-fire: 2.5; three-fires: 2.6 mm) were similar between treatments, and mortality rate was similar between unburned and single fire plots (55% in both). Data were collected in May 2006 to August 2008 in 8-12 plots (10 × 40 m) in each of three treatment areas: control (unburned), single fire (prescribed fire in 2003) and three fires (prescribed fires in 2003, 2004, and 2006). Treatment areas were established in 2002 at each of three sites (200-300 ha).

A site comparison study in 1994-2007 in subarctic boreal forest in Yukon Territory, Canada (16) found that **prescribed burning with long intervals**

between burns increased the density of black spruce *Picea mariana* seedlings. The density of seedlings was higher following burns with long intervals between (8.3 seedlings/m^2) than with short intervals between burns and mature forest sites (0.6 seedlings/m^2 in both). Eight 30 × 30 m plots were established within each of three fire history sites: mature forest (previous fire about 77 years ago); long interval between burns (fire in 2005) and short intervals between burns (fire in both 1990/91 and 2005). Seedlings were counted in ten 0.25 m^2 subplots in each plot in 2008, 2009 and 2010.

A before-and-after study in 2003-2005 in temperate coniferous forest in California, USA (17) found **no effect of prescribed fire on the density of conifer seedlings and saplings**. The changes (after minus before) in density were similar between treatments for both seedlings <1.4 m tall (burned: -735; control: -2,303 individuals/ha) and saplings >1.4 m tall and <10 cm diameter at breast height (burned: -222; control: 74). Data were collected in 2003 (before) and 2005 (after) in five plots (0.04 ha) in each of two burned (in 2004) and two control (unburned) treatment units (~1 ha).

(1) Brose, P.H., and Van Lear, D.H. (1998) Responses of hardwood advance regeneration to seasonal prescribed fires in oak-dominated shelterwood stands. *Canadian Journal of Forest Research*, 28, 31-339.

(2) Elliott, K.J., Hendrick, R.L., Major, A.E., Vose, J.M., and Swank, W.T. (1999) Vegetation dynamics after a prescribed fire in the southern Appalachians. *Forest Ecology and Management*, 114, 199-213.

(3) Provencher, L., Herring, B.J., Gordon, D.R., Rodgers, H.L., Tanner, GW., Hardesty, J.L., ;Brennan, L.A., and Litt, A.R. (2001) Longleaf pine and oak responses to hardwood reduction techniques in fire-suppressed sandhills in northwest Florida. *Forest Ecology and Management,* 148, 63-77.

(4) Elliott, K.J., and Vose, J.M. (2005) Effects of understory prescribed burning on shortleaf pine (*Pinus echinata* Mill.)/mixed-hardwood forests. *Journal of the Torrey Botanical Society,* 132, 236-251.

(5) Hutchinson, T.F., Sutherland, E.K., and Yaussy, D.A. (2005) Effects of repeated prescribed fires on the structure, composition, and regeneration of mixed-oak forests in Ohio. *Forest Ecology and Management,* 24, 210-228.

(6) Wang, G.G., Van Lear, D.H., and Bauerle, W.L. (2005) Effects of prescribed fires on first-year establishment of white oak (*Quercus alba* L.) seedlings in the Upper Piedmont of South Carolina, USA. *Forest Ecology and Management,* 213, 328-337.

(7) Albrecht, M.A., and McCarthy, B.C. (2006) Effects of prescribed fire and thinning on tree recruitment patterns in central hardwood forests. *Forest Ecology and Management,* 226, 88-103.

(8) Alexander, H.D., Arthur, M.A., Loftis, D.L.., and Green, S.R. (2008) Survival and growth of upland oak and co-occurring competitor seedlings following single and repeated prescribed fires. *Forest Ecology and Management,* 256, 1021-1030.

(9) Allen, E.A., Chambers, J.C., and Nowak, R.S. (2008) Effects of a spring prescribed burn on the soil seed bank in sagebrush steppe exhibiting pinyon-juniper expansion. *Western North American Naturalist*, 68, 265-277.

(10) Lombardo, J.A., and McCarthy, B.C. (2008) Silvicultural treatment effects on oak seed production and predation by acorn weevils in southeastern Ohio. *Forest Ecology and Management* 225, 2566-2576.

(11) Moghaddas, J.J., York, R.A., and Stephens, S.L. (2008) Initial response of conifer and California black oak seedlings following fuel reduction activities in a Sierra Nevada mixed conifer forest. *Forest Ecology and Management*, 255, 3141-3150.

(12) Prévosto, B., and Ripert, C. (2008) Regeneration of *Pinus halepensis* stands after partial cutting in southern France: Impacts of different ground vegetation, soil and logging slash treatments. *Forest Ecology and Management*, 256, 2058-2064.

(13) Elliott, K.J., and Vose, J.M. (2010) Short-term effects of prescribed fire on mixed oak forests in the southern Appalachians: Vegetation response. *Journal of the Torrey Botanical Society,* 137, 49-66.
(14) Fiedler, C.E., Metlen, K.L., and Dodson, E.K. (2010) Restoration treatment effects on stand structure, tree growth, and fire hazard in a ponderosa pine/douglas-fir forest in Montana. *Forest Science,* 56, 18-31.
(15) Royse, J., Arthur, M.A., Schörgendorfer, A., and Loftis, D.L. (2010) Establishment and growth of oak (*Quercus alba*, Quercus prinus) seedlings in burned and fire-excluded upland forests on the Cumberland Plateau. *Forest Ecology and Management,* 260, 502-510.
(16) Brown, C.D., Johnstone, J.F. (2012) Once burned, twice shy: Repeat fires reduce seed availability and alter substrate constraints on *Picea mariana* regeneration. *Forest Ecology and Management,* 266, 34-41.
(17) Walker, R.F., Fecko, R.M., Frederick, W.B., Johnson, D.W., and Miller, W.W. (2012) Seedling recruitment and sapling retention following thinning, chipping, and prescribed fire in mixed Sierra Nevada conifer. *Journal of Sustainable Forestry*, 31, 747-776.

6.1.3. Use prescribed fire: effects on understory plants

- Eight of 22 studies (including seven replicated, randomized, controlled studies) from the USA[3,4,12,13,18,20,21,23,24,25,27,28,29,30,31,32,34,36,37], Australia[5,35] and Canada[9] found that prescribed fire increased the **cover**[12,13,24,27,30,31], **density**[35] and **biomass**[3] of understory plants. Six of the studies found it decreased plant cover[4,5,25,29,32,36].Eight found no effect or mixed effects on cover[9,18,23,28,37] and density[20,21,34] of understory plants.
- Fourteen of 24 studies (including 10 replicated, randomized, controlled studies) from the USA[1,2,4,6,8,10,11,13-20,27,30,34,37,] Australia[5,7,26,35], France[33] and West Africa[22] found that prescribed fire increased **species richness**[1,2,5,6,7,11,13,14,15,17,30,33,35] and **diversity**[4,5,30] of understory plants. One[34] study found that it decreased species richness. Nine found no effect or mixed effects on species richness[8,10,16,18,19,20,22,26,37]and diversity[16,19,22] of understory plants.

A replicated, controlled study in 1957-1964 in temperate coniferous forest in Minnesota, USA (1) found that **annual and biannual spring but not summer prescribed fires increased the number of regenerating hazel *Corylus spp.* stems**. Numbers of stems was higher in annual and biannual spring (annual: 38,445; biannual: 30,109/ha) than annual and biannual summer (annual: 3,845; biannual: 13,840) and unburned plots (8,741/ha). The density in single burn plots (single spring: 16,349; single summer: 16,956) was similar to all other treatments. Four plots (~2.5 ha) of each of seven treatments were established in 1957-1960: spring and summer fires carried out: annually, biennially or just once (single) and control (unburned). Data were collected four years after the beginning of the treatment in each plot.

A replicated, controlled study in 1965-1979 in temperate woodland in Minnesota, USA (2) found that **annual prescribed burns increased understory plant species richness**. Numbers of species was higher in burned (25/100 m^2) than unburned plots (13/100 m^2). Data were collected in 1979 in 10 plots (100 m^2) in a 10 ha control (unburned) area and in 11 similar-size plots in an 11 ha burned area (annual prescribed burns 1965-1978).

A controlled study in 1991-1997 in temperate coniferous forest in Louisiana, USA (3) found that prescribed burning increased herbaceous biomass. **Annual herbs productivity was higher in burned** (780-1220 kg dried matter/ha) **than**

in unburned plots (452-472). Data were collected in four replicates of 0.16 ha prescribed burned (in March 1991, February 1994 and March 1997) and control unburned treatment plots. Herbaceous biomass was sampled in July 1997 in three quadrats (0.02 m^2) within each plot.

A before-and-after trial in 1994-1995 in temperate mixed forest in North Carolina, USA (4) found that **prescribed fire increased herbaceous species diversity, and decreased herbs cover**. Cover of the herbaceous layer (all herbaceous species plus woody stems <1.0 cm basal diameter) decreased (before: 36; after: 11%) while diversity of the herb-lair increased (Shannon's index before: 1.01; after: 2.14). Data were collected before (1994) and after prescribed-burning (1995) in six plots (15 × 15 m) and in additional 20 plots (10 × 10 m) after burning.

A replicated, controlled study in 1994-1996 in Mediterranean jarrah *Eucalyptus marginata* forest in Western Australia (5) found that **prescribed fire in restored forest sites increased plant species richness and density, but decreased plant cover**. The density of all plants was higher in burned (35/m^2) and native forest (32/m^2) than unburned plots (6/m^2), while their cover was lower in burned (10%) than unburned plots (48%) and native forest (60%). Weed density was higher in burned (6/m^2) than unburned plots (3/m^2) and native forest (2/m^2). The number of native plant species was higher in burned (40/80 m^2) than unburned plots (28/80 m^2) and the highest in native forest (64/80 m^2), while plant diversity was lower in unburned and burned plots (Shannon's index: 2.3 in both) than native forest (3.2). Data were collected in 1995-1996 in 5-10 burned (prescribed fire in 1994-1995) and five unburned plots (20 × 20 m) in each of three bauxite mine sites rehabilitated in 1981, 1982 and 1983, and in 10 similar size plots in native forest sites (control).

A replicated, randomized, controlled study in 1995-1998 in temperate mixed forest in Florida, USA (6) found that **prescribed fire increased plant species richness in fire-suppressed areas**. Species richness was higher in burned (47-50/400 m^2) than unburned plots (44/400 m^2). Data were collected in 1998 in 32 subplots (400 m^2) in each of one unburned control and three burned 81 ha plots (burned in 1995) replicated in six blocks. All plots had been fire-suppressed for several decades before treatments.

A site comparison study in 2000 in a Mediterranean jarrah *Eucalyptus marginata* forest in Western Australia (7) found that **prescribed burning increased the abundance of native plants at large and small scales and their species richness only at small scale.** The number of native plant species at the small scale was higher in burnt than in unburned plots (burned: 12-13; unburned: 10/m^2), while the number of species at the larger scale was similar between treatments (burned: 57; unburned: 51/30 m^2). Native plant abundance at the small scale (burned: 38-39; unburned: 29/m^2) and large scale (burned: 1,138-1,172; unburned: 876/30 m^2) was higher in burned than unburned plots. Data were collected in 2000 using five lines of 30 quadrats (1 × 1 m) placed in burned sites (prescribed burn in 1996) and 10 lines of 30 similar size quadrats placed in control sites (unburned since 1986). All lines were located in an 11,000 ha study area.

A replicated, controlled study in 1995-1998 in temperate mixed forest in Kentucky, USA (8) found **no effect of prescribed fire on understory plant species richness.** The number of species was similar in burned (32/25 m^2) and unburned plots (26/25 m^2). Data were collected in 1998 in 6-8 replicates of burned (prescribed burned in 1995) and control (unburned) plots (25 m^2) at each of three sites.

A replicated, controlled study in 1999-2000 in boreal forest in Alberta, Canada (9) found that **prescribed burning increased the cover of fireweed *Epilobium angustifolium* but not the cover of cranberry *Viburnum edule*, herbs or *populus spp***. The cover of fireweed was higher in burned plots (burned: 18%; unburned: 4%), while the cover of cranberry (burned: 2%; unburned: 2%), herbs (burned: 6%; unburned: 7%) and populus spp. (burned: 4%; unburned: 6%), as well as the density of *populus spp.* root-suckers (burned: 25,400; unburned: 35,500/ha) were similar between treatments. Four burned (in May 1999) and four unburned 2 x 2 m plots were established within each of six 10 ha forest units. Cover of herbaceous plants was visually estimated in late July 1999. Cover of fireweed, cranberry and *Populus spp*., as well as the root sucker density of *Populus spp*. was evaluated in August 2000.

A replicated, before-and-after study in 1993-2001 in sand pine scrub in Florida, USA (10) found **no effect of prescribed fire on plant species richness**. The total number of plant species, as well as numbers of herbaceous and woody species was similar before and after the fire (10, 3 and 7 species/50 m transect respectively). Plants were monitored along six 50 m transects randomly established inside a 12 ha study site, before (1993) and after (2001) prescribed burning (in May 1993).

A replicated, randomized, controlled study in 1998–2000 in temperate coniferous forest in South Dakota, USA (11) found that **prescribed fire increased plant species richness**. Species richness was higher in burned (8/0.25 m^2) than unburned plots (3/0.25 m^2). Data were collected in July 2000 in 30 plots (0.25 m^2) in each of three replicates of control (unburned) and burned (in May 1999) treatments (45 × 45 m).

A replicated, controlled study in 2001-2003 in temperate mixed forest in Georgia and Tennessee, USA (12) found that **prescribed burning increased the cover of understory plants but did not affect plant species richness**. The cover of herbaceous plants and tree seedlings <50 cm tall was higher in burned sites (burned: 26%; unburned: 24%) while numbers of species was similar between treatments (burned: 34.2 unburned: 34.0). Data were collected in 2002 in five 10 × 20 m plots in each of four burned (prescribed burn in March 2001) and two control unburned sites (total of 30 plots).

A site comparison study in 1998-2001 in temperate coniferous forest in Arizona, USA (13) found that **prescribed burning increased plant species richness and cover.** Total species richness of plants (burned: 40; unburned: 24 species/0.1 ha) and total plant cover (burned: 50%; unburned: 11%) were higher in burned plots. Herbaceous species richness (burned: 33; unburned: 19) as well as exotic species richness (burned: 2; unburned: 0) and cover (burned: 1%; unburned: 0%) were higher in burned plots. In contrast, there was no difference between treatments for woody species richness (burned: 6; unburned: 6) or plant species diversity (Shannon's index burned: 1.9; unburned: 1.4). Data were collected in 2001 in 30 plots (0.1 ha) within each of two areas (270 ha): burned (severely burned in 1993) and unburned.

A replicated, controlled study in 1994-1999 in temperate oak forest in Ohio, USA (14) found that **prescribed burning increased plant species richness on**

a small scale, but not on a larger-scale. Species richness of all plants within 2 m^2 plots was higher in burned plots (burned: 18; unburned: 16/2 m^2). Species richness of annual forbs (burned: >1; unburned: <1/2 m^2), summer-flowering forbs (burned: 4; unburned: 32/ m^2), grasses (burned: >1; unburned: <1/2 m^2) and woody seed-banking species (burned: >1; unburned: <1/2 m^2) was higher in burned plots. Species richness of shade-tolerant tree seedlings (burned: <2; unburned: >2/2 m^2) and oak–hickory tree seedlings (burned: 1; unburned: >1/2 m^2) was higher in unburned plots. Species richness of spring-flowering forbs (4-5/2 m^2), sedges (1/2 m^2) and shrubs (2/2 m^2) was similar between treatments. Species richness within larger 1,250 m^2 plots was similar between treatments (burned: 67; unburned: 63/1,250 m^2). Two burned (annually 1996-1999 and twice in 1996 and 1999) and one unburned treatment units (25 ha) were replicated at four sites. Data were collected in 1999 using 16 quadrats (2 m^2) in each of nine plots (1,250 m^2) within each treatment unit.

A replicated, randomized, controlled study in 2001-2004 in temperate coniferous forest in Montana, USA (15) found that **prescribed fire increased the number of understory exotic species and forbs at the large plot scale and decreased the number of native species at smaller scale**. Plot-scale species richness was higher in burned plots for exotic plants (unburned: 4; burned: 6/1,000 m^2) and forbs (unburned: 34; burned: 38/1,000 m^2). On a smaller scale species richness for native species was lower in burned plots (unburned: 11; burned: 10/m^2). For all plants together, plot-scale and a smaller scale were similar between treatments for species richness (plot-scale: 57 - 60/1,000 m^2; small-scale: 11/m^2) and cover (both scales: 26-28%). Three replicates of burned and unburned control treatments (9 ha) were established in 2001. Species composition was determined in 2004 in 12 quadrats (1 m^2) in each of 10 plots (1,000 m^2) within each treatment (total of 1,440 quadrats).

A replicated, randomized, controlled study in 2000-2004 in temperate coniferous forest in Oregon, USA (16) found **no effect of prescribed fire on understory species richness and diversity**. Number of species/400 lm^2 plot (burned: 29; control: 30) and Shannon's index of diversity (burned: 0.11; control: 0.12) were similar between treatments. Data were collected in 2004 in 10-30 plots (400 m^2) in each of four burned (prescribed burnt in 2000) and four control (unburned) experimental units.

A replicated, randomized, controlled study in 2001-2004 in temperate coniferous forest in Montana, USA (17) found that **prescribed burns increased species richness of uncommon but not of common plant species**. The number of uncommon species was higher in burned plots (burned: 14; unburned: 11) while the number of common species was similar between treatments (burned: 30; unburned: 32). Data were collected in 2004 in 10 burned (prescribed burn in 2002) and 10 control (unburned) treatment plots (1,000 m^2) in each of three blocks.

A replicated, randomized, controlled study in 2001-2004 in temperate coniferous forest in California, USA (18) found **no effect of prescribed fire on cover and species richness of understory vegetation**. Total plant cover increased by 0.6% in unburned compared with 0.8% in early- and 2.0% in late-burned plots. Numbers of species/1 m^2 increased by 0.12 in unburned compared with a 0.34 increase in early and a 0.03 decrease in late burned plots. Numbers of species/0.1 ha decreased by 0.7 in unburned compared with increases of 6.0 in early- and 6.6 in late-burned plots. Data were collected in 2004 in 10 plots (0.1 ha) that were established in each of three unburned, three early-burned (June 2002) and three late-burned (September-October 2001) randomly assigned treatment units (15-20 ha).

A replicated, randomized, controlled study in 2000-2004 in temperate broadleaf forest in Ohio, USA (19) found **no effect of burning on soil seed-bank species richness or diversity**. The total number of species (burned: 43; unburned: 38/1000 cm^3 soil) as well as diversity (Shannon's index burned: 3.23; unburned: 3.11) were similar between treatments. Ten plots (20 × 50 m) were established within each burned (prescribed burned in spring 2001) and control (unburned) treatments (20 ha) at each of two sites. Species richness and diversity were determined by monitoring emerging seeds in 10 soil samples (1,000 cm^3) extracted from each plot in summer 2004.

A replicated, randomized, controlled study in 2000-2003 in temperate mixed forest in California, USA (20) found **no effect of prescribed burning on understory plant species richness and cover**. Numbers of species (burned: 3; unburned: 4/10 m^2) and cover (burned: 9%; unburned: 8%) were similar between treatments. Data were collected in 2003 in 9-49 plots (10 m^2) in each of three control (unburned) and three burned (prescribed fire in 2001) treatments units (4 ha).

A replicated, controlled before-and-after study in 2000-2004 in Piedmont forest in South Carolina, USA (21) found that **prescribed burning decreased the density of tree saplings, but increased the density of seedlings and other plants species richness and cover**. The density of tree saplings >1.4 m tall and <10 cm diameter at breast height decreased in burned plots (-175/ha) whereas it increased in unburned plots (243/ha). Increases in tree seedlings <1.4 m tall (burned: 17,850; unburned: 8,550/ha), the cover of vines (burned: 3.2%; unburned: -2.7%), herbaceous species (burned: 2.6%; unburned: -0.2%) and grasses (burned: 3.3%; unburned: -0.5%) and the number of plant species (burned: 41; unburned: 32/0.1 ha) were greater in burned plots. Declines in the cover of shrubs were similar between treatments (burned: -0.03%; unburned: -0.41%). Ten plots (0.1 ha) were established in each of three control (unburned) and three burned (in 2001-2002) treatment units. Data were collected three years after treatment.

A replicated, randomized, controlled study in 1994-2003 in savanna woodland in West Africa (22) found **no effect of annual prescribed fire on herbaceous species richness or diversity**. Numbers of species (unburned: 13-16; burned: 14-16/0.25 ha) and diversity (Shannon's index unburned: 2.4-2.8; burned: 2.7-2.9) was similar between treatments. Data were collected in 2003 in two control (unburned) and two burned treatment plots (0.25 ha) replicated in eight blocks, at each of two sites (18 ha). Annual prescribed fires were carried out at the end of the rainy season in 1994-2003.

A replicated, randomized, controlled study in 2000-2007 in temperate broadleaf forest in North Carolina and Ohio, USA (23) **found that prescribed burning had mixed effects on the cover of different plant groups at two different sites**. At the 'cool temperate climate' site the number of hardwood tree saplings (>1.4 m tall) (burned: 430/ha, unburned: 370/ha), cover of herbs (burned: 20%, unburned: 13%) and shrubs and tree seedlings (< 1.4 m tall) (burned: 50%, unburned: 25%) were higher in burned plots. At the same site, the cover of shrubs (>1.4 m tall) was lower in burned plots (burned: 2%, unburned: 8%). At the 'warm continental climate' site, the number of tree saplings (burned: 900/ha, unburned: 1,800/ha) and cover of shrubs (>1.4 m tall) (burned: 9%, unburned: 28%) were higher in unburned plots, while the cover of tree seedlings (burned: 8%, unburned: 6%) was higher in burned plots. At the 'cool temperate climate' site, cover of herbaceous species (burned: 4%, unburned: 5%) and shrub seedlings (burned: 8%, control: 11%) was similar between treatments. Three pairs of burned (in 2002-2003) and control (unburned) treatment units (10-26 ha) were established at each of a 'cool temperate climate' and 'warm continental climate' site. Data were collected 3-4 years post-treatments in 10 plots (0.1 ha) in each treatment unit.

A replicated, randomized, controlled study in 1997-2006 in temperate coniferous forest in Oregon, USA (24) found that **winter burning of cut western juniper *Juniperus occidentalis* trees increased annual herbaceous plant cover**. Cover of all herbaceous plants and cover of perennial grasses was higher in 1st-year burn (30% and 22% respectively) and 2nd-year burn plots (28% and 18%) than in unburned plots (18% and 8%). Cover of annual herbaceous species and of Sandberg’s bluegrass *Poa secunda* was higher in 1st-year burn (5% and 2% respectively) than in unburned plots (1% and 0%) and intermediate in 2nd-year burn plots (4% and 1%). Cover of cheatgrass *Bromus tectorum* was higher in unburned (10%) than in 1st-year and 2nd-year burn plots (2% in both), while cover of perennial forbs was similar (<1%) in all treatments. Three treatments (0.5 ha) were randomly assigned to each of five blocks in which all juniper trees were cut down in 1997. Treatments were: unburned, 1st-year and 2nd-year burned (cut trees burned the first and second winter after cutting respectively). In 2006 herbaceous cover was measured in four 0.2 m^2 quadrats under each of 10 cut trees in each treatment.

A replicated, controlled study in 1998-2006 in temperate forest in Louisiana, USA (25) found that **prescribed fire decreased understory vegetation cover**. The total cover of understory vegetation was lower in burned treatments than in unburned plots (burned: 58-62%; unburned 68%). Data were collected in 2006 in three plots (0.07 ha) of each of March, May and July burns (prescribed burn in 1999, 2001, 2003, and 2005), and control (untreated since 1998) treatments. Each of the 12 plots was planted with 196 longleaf pine seedlings in 1993-1994.

A replicated, before-and-after study in 2005-2009 in Mediterranean type shrubland in Western Australia (26) found that **prescribed fire increased plant species richness in natural sites, but decreased species richness in restored mine-sites**. Plant species richness increased in natural areas after fire (pre-fire: 99; post-fire: 116) and decreased after fire in restored areas (pre-fire: 118; post-fire: 80). The percentage of species that persisted after fire was lower in restored (50%) than in natural areas (91%). Prescribed fire was applied in 2005-2007 to a 40 × 40 m plot at each of three restored mine-sites (8-24 years before the experiment) and five natural sites. Data were collected before (2005) and two years after fire.

A controlled study in 2004-2006 in temperate mixed forest in North Carolina and Georgia, USA (27) found that **prescribed fire increased the cover of herbaceous plants only in one out of three sites**. The cover of herbaceous plants was higher in burned than unburned plots at one site (burned: 132; unburned: 88%) and similar between treatments at the second (burned: 55;

unburned: 37%) and the third sites (burned: 2; unburned: 1%). Data were collected in 2006 in 10-12 plots (10 × 20 m) in a burned area (in 2004) and in 4-6 plots in an adjacent unburned area (control) in each site.

A replicated, controlled study in 2004-2008 in temperate broadleaf forest in Pennsylvania, USA (28) found that **prescribed fire decreased fruit production but not the cover of some herbaceous species**. Four years after treatment, the total number of fruit/plot for three herbaceous species: painted trillium *Trillium undulatum*, sessile bellwort *Uvularia sessilifolia*, and Indian cucumber root *Medeola virginiana* was lower in burned plots (burned: 0-380; unburned: 0-430), while their relative cover (0-3%) was similar between treatments. Cover of bramble *Rubus spp.* (1-25%) and hay-scented fern *Dennstaedtia punctilobula* (0-70%) and the number of tree saplings (0.0-1.8/m^2) were similar between treatments. Data were collected in 2008 in three blocks of 12 burned (prescribed fire on May 2004) and 12 unburned plots (50 × 80 m).

A replicated, randomized, controlled study in 2002-2005 in temperate coniferous forest in Alabama, USA (29) found that prescribed fire **decreased the density of understory shrubs and trees and increased the cover of grasses**. The density of small hardwoods (<3 cm diameter at breast height) (burned: ~300; unburned: >1,500 trees/ha) and cover of tall shrubs (>1.4 m) (burned: 10%; unburned: 33%) were higher in unburned plots. The cover of grasses was higher in burned plots (burned: 20%; unburned: 7%) and the cover of short shrubs (<1.4 m) (45-57%) and forbs (3-10%) was similar between treatments. Control (unburned) and burned (prescribed burned in 2002 and 2004) treatment units were replicated in three blocks. Data were collected in 2005 in ten 20 × 50 m subplots within each treatment unit.

A replicated, controlled study in 2000-2006 in temperate broadleaf forest in West Virginia, USA (30) found that **prescribed fire increased understory vegetation cover and diversity**. Cover (burned: 28%; unburned: 7%), species richness (burned: 2.9; unburned: 1.5 species/m^2) and diversity (Shannon's index burned: 1.31; unburned: 1.06) were higher in burned plots. Eight burned (in 2001) and eight control (unburned) treatment plots (20 × 20 m) were established in each of four sites. Data were collected in 2006 in five quadrats (1 m^2) in each plot.

A replicated, randomized, controlled study in 1976-2008 in temperate coniferous forest in Arizona, USA (31) found that **frequent prescribed burning increased the cover of grasses and specifically wheatgrasses, but not of forbs and of total herbaceous plants**. At one site, cover of grasses was higher with 1-year interval burns than in unburned plots (1-year: 8%; 2-, 4-, 6-, 8- and 10-years: 5-8%; unburned: 5%) and cover of wheatgrasses was higher with 1-, 2- and 4-year intervals than in unburned plots (1-, 2- and 4-years: 4-5%; 6-, 8- and 10-years: 2-3%; unburned: <1%). At that same site, cover did not differ between treatments for forbs (all burn intervals: 3-6%; unburned: 2%) or total herbaceous vegetation (all burn intervals: 9-12%; unburned: 5%). At a second site the cover of grasses (all burn intervals 3-7%; unburned: 3%), wheatgrasses (all burn intervals: <1%; unburned: 0%), forbs (all burn intervals: 1-3%; unburned: 2%) and total herbaceous cover (all burn intervals: 4-10%; unburned: 5%) were similar between treatments. Data were collected in 2007-2008 in three plots of each of 1-, 2-, 4-, 6-, 8- and 10-year intervals between prescribed fires and control (unburned >75 years) treatment plots (100 × 100 m) established in 1976-1977 in each of two sites.

A replicated, before-and-after study in 2006-2010 in temperate coniferous forest in Washington State, USA (32) found that **prescribed burning decreased the cover of understory vegetation**. Understory vegetation cover was lower 3 years post-fire (pre-fire: 107%; post-fire: 40%). Data were collected in 2006 (pre-fire) and in 2010 (post-fire) in five 20 x 20 m plots that were prescribed burned in 2007.

A replicated, randomized, controlled study in 2005 in Mediterranean Aleppo pine *Pinus halepensis* woodland in France (33) found that **prescribed burning increased plant species richness and diversity and decreased shrub cover but did not affect the cover of herbaceous plants**. Numbers of species (unburned: 27; burned: 33/plot) and diversity (Shannon's index unburned: 3.2; burned: 3.7) were higher in burned plots, while shrub cover was lower in burned plots (unburned: 40%; burned: 29%). Herbaceous plant cover was similar between treatments (unburned: 24%; burned: 30%). Data were collected in 2009 in eight unburned control and eight burned (prescribed fire in 2005) plots (14 × 14 m). All plots were thinned in 2004 (from 410 to 210 trees/ha).

A replicated, controlled study in 1946-2012 in temperate broadleaf forest in North Carolina, USA (34) found that **prescribed fire decreased the soil seed-bank species richness but not species diversity or density**. Numbers of emerged species was lower after prescribed fire (before: 12.1-12.3; after: 11.0-11.3/m^2), while diversity (Shannon's index in m^2 1.9-2.1) and total density (419-603 emergents/m^2) were similar between treatments. Five plots (5 ha) were burned in 2009-2010. Data were collected using four 0.06 m^2 soil seed-bank samples, two before and two after burning, taken from each of six subplots (0.5 ha) within each plot.

A replicated, controlled study in 2001-2005 in temperate eucalyptus forest in Queensland, Australia (35) found that **prescribed fire increased plant density and species richness**. The density of <1 m tall native plants (burned: 13; unburned: 9 individuals/6 m^2), ferns (burned: 2-3; unburned: 2) and resprouters (burned: 12-13; unburned: 9) and of 1-3 m tall native plants (burned: 3; unburned: 2) and resprouters (burned: 3; unburned: 2) was higher in burned than unburned plots. The density of other <7.5 m tall plant groups examined was not affected by burning. Data were collected in 2009 in three subplots (6 × 1 m) in each of four burned (prescribed burn with 2 or 4 year intervals since 1971) and two control (no fires since 1969) 0.08 ha plots.

A before-and-after study in 2003-2005 in temperate coniferous forest in California, USA (36) found that **prescribed fire decreased the cover of understory vegetation**. The change (after minus before) in understory vegetation cover was more negative in burned (-19%) than unburned plots (-2%). Data were collected in 2003 (before) and 2005 (after) in five plots (0.04 ha) in each of two burned (in June 2004) and two control (unburned) treatment units of approximately 1 ha.

A replicated, controlled study in 2004-2011 in temperate coniferous forest in Arizona, USA (37) found **no effect of burning on plant cover or on species richness**. Total plant cover (3-5%), species richness (33-37 species) and diversity (Simpson's index 0.8-0.9) were similar between treatments. Four burned (prescribed burn in 2006) and four unburned treatment units (1 ha) were replicated in six blocks. Data were collected in 2011 in one 0.04 ha plot in each treatment unit (total of 48 plots).

(1) Buckman, R.E. (1964) Effects of prescribed burning on hazel in Minnesota. *Ecology,* 45, 626-629.

(2) White, A.S. (1983) The effects of thirteen years of annual prescribed burning on a *quercus ellipsoidalis* community in Minnesota. *Ecology,* 64, 1081-1085.

(3) Haywood, J.D., Tiarks, A.E., Elliott-Smith, M.L., and Pearson, H.A. (1998) Response of direct seeded Pinus palustris and herbaceous vegetation to fertilization, burning, and pine straw harvesting. *Biomass and Bioenergy,* 14, 157-167.

(4) Elliott, K.J., Hendrick, R.L., Major, A.E., Vose, J.M., and Swank, W.T. (1999) Vegetation dynamics after a prescribed fire in the southern Appalachians. *Forest Ecology and Management*, 114, 199-213.

(5) Grant, C.D., and Loneragan, W.A. (1999) The effects of burning on the understory composition of 11-13 year-old rehabilitated bauxite mines in Western Australia. *Plant Ecology*, 145, 291-305.

(6) Provencher, L., Herring, B.J., Gordon, D.R., Rodgers, H.L., Galley, K.E.M., Tanner, G.W., Hardesty, J.L., and Brennan L.A. (2001) Effects of hardwood reduction techniques on longleaf pine sandhill vegetation in northwest Florida. *Forestry Chronicle,* 84, 70-75.

(7) Burrows, N.D., Ward, B., and Cranfield, R. (2002) Short-term impacts of logging on understory vegetation in a jarrah forest. *Australian Forestry,* 65, 47-58.

(8) Kuddes-Fischer, L.M., and Arthur, M.A. (2002) Response of understory vegetation and tree regeneration to a single prescribed fire in oak-pine forests. *Natural Areas Journal*, 22, 43-52.

(9) Frey, B.R., Lieffers, V.J., Munson, A.D., and Blenis, P.V. (2003) The influence of partial harvesting and forest floor disturbance on nutrient availability and understory vegetation in boreal mixedwoods. *Canadian Journal of Forest Research,* 33, 1180-1188.

(10) Greenberg, C.H. (2003) Vegetation recovery and stand structure following a prescribed stand-replacement burn in sand pine scrub. *Natural Areas Journal*, 23, 411-151.

(11) Wienk, C.L., Sieg, C.H., and McPherson, G.R. (2004) Evaluating the role of cutting treatments, fire and soil seed banks in an experimental framework in ponderosa pine forests of the Black Hills, South Dakota. *Forest Ecology and Management,* 192, 375-393.

(12) Elliott, K.J., and Vose, J.M. (2005) Effects of understory prescribed burning on shortleaf pine (*Pinus echinata* Mill.)/mixed-hardwood forests. *Journal of the Torrey Botanical Society,* 132, 236-251.

(13) Huisinga, K.D., Laughlin, D.C., Fulé, P.Z., Springer, J.D., and McGlone, C.M. (2005) Effects of an intense prescribed fire on understory vegetation in a mixed conifer forest. *Journal of the Torrey Botanical Society,* 132, 590-601.

(14) Hutchinson, T.F., Boerner, R.E.J., Sutherland, S., Sutherland, E.K., Ortt, M., and Iverson, L.R. (2005) Prescribed fire effects on the herbaceous layer of mixed-oak forests. *Canadian Journal of Forest Research,* 35, 877-890.

(15) Metlen, K.L. and Fiedler, C.E. (2006) Restoration treatment effects on the understory of ponderosa pine/Douglas-fir forests in western Montana, USA. *Forest Ecology and Management,* 222, 355-369.

(16) Youngblood, A., Metlen, K.L., and Coe, K. (2006) Changes in stand structure and composition after restoration treatments in low elevation dry forests of northeastern Oregon. *Forest Ecology and Management,* 234, 143-163.

(17) Dodson, E.K., Metlen, K.L., and Fiedler, C.E. (2007) Common and uncommon understory species differentially respond to restoration treatments in ponderosa pine/Douglas-fir forests, Montana. *Restoration Ecology,* 15, 696-708.

(18) Knapp, E.E., Schwilk, D.W., Kane, J.M., and Keeley, J.E. (2007) Role of burning season on initial understory vegetation response to prescribed fire in a mixed conifer forest. *Canadian Journal of Forest Research,* 37, 11-22.

(19) Schelling, L.R., and McCarthy, B.C. (2007) Effects of prescribed fire and thinning on the spatial heterogeneity of the seed bank in a mixed oak forest. *Natural Areas Journal,* 27, 320-331.

(20) Wayman, R.B., and North, M. (2007) Initial response of a mixed-conifer understory plant community to burning and thinning restoration treatments. *Forest Ecology and Management*, 239, 32-44.

(21) Phillips, R.J., and Waldrop, T.A. (2008) Changes in vegetation structure and composition in response to fuel reduction treatments in the South Carolina Piedmont. *Forest Ecology and Management,* 255, 3107-3116.
(22) Savadogo, P., Tiveau, D., Sawadogo, L., and Tigabu, M. (2008) Herbaceous species responses to long-term effects of prescribed fire, grazing and selective tree cutting in the savanna-woodlands of West Africa. *Perspectives in Plant Ecology, Evolution and Systematics,* 10, 179-195.
(23) Waldrop, T.A., Yaussy, D.A., Phillips, R.J., Hutchinson, T. A., Brudnak, L., and Boerner, R.E.J. (2008) Fuel reduction treatments affect stand structure of hardwood forests in Western North Carolina and Southern Ohio, USA. *Forest Ecology and Management,* 225, 3117-3129.
(24) Bates, J.D., and Svejcar, T.J. (2009) Herbaceous succession after burning of cut western juniper trees. *Western North American Naturalist,* 69, 9-25.
(25) Haywood, J. D. (2009) Eight years of seasonal burning and herbicidal brush control influence sapling longleaf pine growth, understory vegetation, and the outcome of an ensuing wildfire. *Forest Ecology and Management,* 258, 295-305.
(26) Herath, D.N., Lamont, B.B., Enright, N.J., and Miller, B.P. (2009) Impact of fire on plant-species persistence in post-mine restored and natural shrubland communities in southwestern Australia. *Biological Conservation,* 142, 2175–2180.
(27) Elliott, K.J., and Vose, J.M. (2010) Short-term effects of prescribed fire on mixed oak forests in the southern Appalachians: Vegetation response. *Journal of the Torrey Botanical Society,* 137, 49-66.
(28) Huebner, C.D., Gottschalk, K.W., Miller, G.W., and Brose, P.H. (2010) Restoration of three forest herbs in the Liliaceae family by manipulating deer herbivory and overstorey and understory vegetation. *Plant Ecology and Diversity*, 3, 259-272.
(29) Outcalt, K.W., and Brockway, D.G. (2010) Structure and composition changes following restoration treatments of longleaf pine forests on the Gulf Coastal Plain of Alabama. *Forest Ecology and Management*, 259, 1615-1623.
(30) Royo, A.A., Collins, R., Adams, M.B., Kirschbaum, C., and Carson, W.P. (2010) Pervasive interactions between ungulate browsers and disturbance regimes promote temperate forest herbaceous diversity. *Ecology,* 91, 93-105.
(31) Scudieri, C.A., Sieg, C.H., Haase, S.M., Thode, A.E., and Sackett, S.S. (2010) Understory vegetation response after 30 years of interval prescribed burning in two ponderosa pine sites in northern Arizona, USA. *Forest Ecology and Management,* 260, 2134-2142.
(32) Fonda, R.W., and Binney, E.P. (2011) Vegetation response to prescribed fire in Douglas-fir forests, Olympic National Park. *Northwest Science,* 85, 30-40.
(33) Prévosto, B., Bousquet-Mélou, A., Ripert, C., and Fernandez, C. (2011) Effects of different site preparation treatments on species diversity, composition, and plant traits in *Pinus halepensis* woodlands. *Plant Ecology*, 212, 627-638.
(34) Keyser, T.L., Roof, T., Adams, J.L., Simon, D., and Warburton, G. (2012) Effects of prescribed fire on the buried seed bank in mixed-hardwood forests of the southern appalachian mountains. *Southeastern Naturalist*, 11, 669-688.
(35) Lewis, T., Reif, M., Prendergast, E., and Tran, C. (2012) The effect of long-term repeated burning and fire exclusion on above- and below-ground Blackbutt (*Eucalyptus pilularis*) forest vegetation assemblages. *Austral Ecology,* 37, 767-778.
(36) Walker, R.F., Fecko, R.M., Frederick, W.B., Miller, W.W., and Johnson, D.W. (2012) Influences of thinning, chipping, and fire on understory vegetation in a Sierran mixed conifer stand. *Journal of Sustainable Forestry,* 31, 493-517.
(37) Huffman, D.W., Stoddard, M.T., Springer, J.D., Crouse, J.E., and Chancellor, W.W. (2013) Understory plant community responses to hazardous fuels reduction treatments in pinyon-juniper woodlands of Arizona, USA. *Forest Ecology and Management*, 289, 478-488.

6.2. Use herbicides to remove understory vegetation to reduce wildfires

- We found no evidence for the effects of using herbicides to remove understory vegetation in reducing wildfiress.

6.3. Mechanically remove understory vegetation to reduce wildfires

- We found no evidence for the effects of mechanically removing understory vegetation in reducing wildfiress.

Water management

6.4. Recharge groundwater to restore wetland forest

- We found no evidence for the effects of recharging groundwater in restoring wetland forest.

Background
In human-dominated landscapes groundwater levels may become depleted due to overuse of water, or may be kept artificially low for agricultural purposes. This can cause nearby wetland forests to experience drought. Allowing the groundwater level to rise again could help to restore wetland forests.

6.5. Construct water detention areas to slow water flow and restore riparian forests

- We found no evidence for the effects of constructing water detention areas in slowing water flow and restore riparian forests on forests.

Background
Creating water retention areas may protect downstream urban areas from floods and may be used to restore riparian forests.

6.6. Introduce beavers to impede water flow in forest watercourses

- We found no evidence for the effects of introducing beavers in impeding water flow in forest watercourses on forests.

Background
By constructing dams, beavers can obstruct water flow and can increase the water level. This may alter the species composition of forests in favour of species that can cope with high water levels.

Change disturbance regime

6.7. Use clearcutting to increase understory diversity

- Eight of 12 studies (including three replicated, randomized, controlled studies) in Belgium[8], Brazil[4,21], Canada[11,14,22], China[15], Germany[17], Israel[13], spain[16] and the USA[3,10] found that clearcutting increased the **cover**[10,14,16,22] and **species richness**[3,10,13,15,16,17,22] of **understory plants**. Two found it decreased the density[21] and species richness[11], and two found no effect[4] or mixed effects[8].
- Three of six studies (including five replicated, randomized, controlled studies) in Brazil[9], Canada[2,5,12,19] and Spain[16] found that clearcutting increased the **density**[5,9] and **species richness**[2] of **young trees**. One found it decreased new tree density[16] and two found no effect[19] or mixed effects depending on the tree species[12].
- Three of nine studies (including four replicated, randomized, controlled studies) in Australia[23], Brazil[4], Canada[2,6,11,18], Japan[24] and the USA[1,7] found that clearcutting decreased **density**[4,11], **species richness**[4,11] and **diversity**[1] of mature trees. One study found it increased **trees species richness**[7] .Six studies found no effect or mixed effects on **tree density**[4,11,18], size[4,6] and **species richness**[4,11,23,24] and diversity[24].
- One replicated, randomized, controlled study in Finland[20] found that clearcutting decreased total forest biomass, and particularly the **biomass of evergreen shrubs**.

Background

Clearcutting of native forests is an undesired action from the perspective of nature conservation. However, it can be used as a conservation management practice to restore natural open areas that were artificially planted, to preserve herbaceous and other understory plant diversity, and for changing forest composition whenever desired. Studies comparing the effects of clearcutting with partial thinning are discussed in 'Threat: Biological resource use - Use partial retention harvesting instead of clearcutting'.

A replicated, controlled, before-and-after study in 1992-1993 in bottomland hardwood forest in Texas, USA (1) found that **clearcutting increased tree species diversity**. Shannon's index for trees <4 cm DBH increased in clearcut plots (before: 2.21; after: 2.33), but remained similar in partial-cut (before: 2.30; after: 2.33) and control plots (before: 1.95; after: 1.98). In 1992, three treatment plots (8.1 ha): clearcut (all woody vegetation cut), partial-cut (50% of basal area cut) and control (uncut) were replicated in three blocks. Data were collected in 1993 in nine subplots (40 m^2) in each plot.

A replicated, randomized, controlled study in 1993-1995 in temperate broadleaf forest in Indiana, USA (2) found that cutting treatments **increased species richness of tree saplings but not of small seedlings, and had mixed effects on mature tree species richness**. Species richness (in 100 m^2 plot) of saplings (>1 m tall, <3 cm DBH) was higher in recent clearcut (8.7-9.1) and group cut (7.8-8.5) than in uncut plots (2.3-3.3), but similar to uncut plots for old clearcut ((4.9-6.1), old group cut (3.0-5.0), recent single tree cut (4.7-5.6) and old single tree cut plots (2.1-2.7). Species richness of seedlings (<1 m tall) was similar between treatments (7.2-11.6). Species richness of mature trees (>3 cm DBH) was higher in old clearcut plots (10.4-10.7) and lower in single tree cut plots (recent single tree: 2.7-3.2; old single tree: 2.7-2.8) compared with uncut plots (6.8-7.3), but similar to uncut plots for recent clearcut (7.6-9.6), recent group cut (5.9-6.5) and old group cut (6.0-6.6). Species richness was calculated in a 100 m^2 plot in each of 37 clearcutting, 45 group selection (cutting in groups, 0.1-1.6 ha) and 44 single tree selection cuts (cutting single trees, 0.005-0.012 h), recent (7-15 years) and old (16-27 years) treatment units and in 24 uncut plots (uncut >80 years).

A replicated, randomized, controlled study in 1993-1995 in temperate broadleaf forest in Indiana, USA (3) found that **cutting treatments increased species richness but not the diversity of herbaceous species and low plants**. Species richness (in 4 m^2 quadrat) was higher in clearcut (43) and group cut (47) compared to single tree cut (36) and uncut plots (34). Diversity (Shannon's index) was similar in all treatments (2.6-2.8). One plot (100 m^2) was established in each of 12 clearcut, 16 group selection (cutting in groups, 0.1-1.6 ha), 17 single tree selection cuts (cutting single trees, 0.005-0.012 h) and 12 uncut (>80 years) treatment plots. Data were collected in four quadrats (4 m^2) in each plot

A replicated, randomized, controlled study in 1985-1997 in tropical forest in Pará State, Brazil (4) found that **clearcuttng decreased species richness, density and size of trees, but not species richness, density and size of vines, herbaceous species and grasses**. Species richness of trees (>15 cm diameter at breast height) was higher in uncut (20/plot) and low intensity cuts (20) than in clearcut and moderate intensity cuts plots (clearcutting: 7/plot; moderate-intensity: 12). Numbers of trees/ha was lower in moderate intensity cut plots (164) than in the other treatments (uncut: 323; low intensity: 339; clearcut: 296). Trees basal area (m^2/ha) was higher in low (16) and moderate intensity cuts (17) than in clearcuts (9) and highest in uncut plots (uncut: 25). Density of vines (uncut: 17,468; low intensity: 8,758; moderate intensity: 11,174; clearcuts: 9,268), herbaceous species (uncut: 1,210; low intensity: 1,131; moderate intensity: 1,401; clearcuts: 1,417) and grasses (uncut: 334; low intensity: 223; moderate intensity: 270; clearcuts: 6,362), and species richness/plot of vines (uncut: 13; low intensity: 13; moderate intensity: 12; clearcuts: 12), herbaceous species (uncut: 3; low intensity: 2; moderate intensity 2; clearcuts: 3) and grasses (<1 in all treatments) were similar between treatments. Data were collected in 1996-1997 in four treatments (20 × 70 m^2): uncut, low intensity cuts (trees ≥45 cm diameter at breast height removed), moderate intensity cuts (trees ≤20 and ≥60 cm removed), and clearcutting, replicated randomly in four blocks in 1985.

A replicated, randomized, controlled study in 1998-2001 in boreal forest in Quebec, Canada (5) found that **clearcutting increased the density of new mountain maple *Acer spicatum* stems and the relative regrowth rates of mountain maple and balsam fir *Abies balsamea* compared with thinning**. Density of new mountain maple stems was higher in clearcut (13 stems/m^2), compared to 33% cut (3), 66% cut (3) and uncut plots (5). Mountain maple relative trunk growth (clearcut: 240%; 66% cut: 130%: 33% cut: 140%; uncut: 130%) and mortality rate (clearcut: 35%; 66% cut: 20%; 33% cut: 27%; uncut: 30%) were similar between treatments. Balsam fir relative height growth ratio for stems <1 m differed between the four different treatments (clearcut: 25%:

66% cut: 15%: 33% cut: 9%; uncut: 4%), while those of stems 1-3 m were similar between treatments (17%, 17%, 9% and 7% respectively). There were three replicates (1-2.5 ha) of four treatments: clearcutting, 66% cut, 33% cut and no harvest (100%, 61%, 33% and 0% of basal area removed respectively). Plots were established in winter 1998-1999. Monitoring was in 1 m^2 quadrats (120 for mountain maple, 226 for balsam fir).

A replicated, randomized, controlled study in 1995-1999 in temperate mixed forest in Ontario, Canada (6) found **no effect of clearcutting on the size of trees and shrubs**. There was no difference between treatments for: height of <4.0 m tall (71-140 cm), and DBH of >4 m tall and 1-10 cm DBH (5.7-6.2 cm) coniferous trees, height (59-177 cm) and density (5,417-12,292 stems/ha) of hardwood regenerations, and height (109-168 cm) and ground cover (33-51%) of shrubs. Data were collected in 1999 in four treatment plots (100 × 100 m): 0% (clearcut), 36%, 68%, and 100% (uncut) of basal area removed, established in 1995 in each of four blocks.

A replicated, randomized, controlled study in 1998–2000 in temperate coniferous forest in South Dakota, USA (7) found that **clearcutting increased tree species richness**. Tree species richness/plot was higher in clearcut (15) than partial-cut plots (8) and the lowest in uncut plots (3). Data were collected in July 2000 in six uncut, six partial-cut (retaining 12 m^2/ha basal area) and six clearcut plots (45 × 45 m) established in winter 1998-1999. Three plots of each treatment were prescribed burned in May 1999.

A site comparison study in temperate broadleaf forest in Belgium (8) found that **clearcutting had mixed effects on forest herbaceous species**. Woodland germander *Teucrium scorodonia,* pill sedge *Carex pilulifera,* and common bracken *Pteridium aquilinum* were more frequent in clearcuts (30%, 25% and 20%) than in uncut forest areas (9%, 15% and 7%). Broad buckler-fern *Dryopteris dilatata*, remote sedge *Carex remota*, enchanter's-nightshade *Circaea lutetiana*, hairy wood-rush *Luzula pilosa* and greater wood-rush *L. sylvatica* were more frequent in uncut forest areas (62%, 60% 43%, 12% and 10%) than in clearcuts (46%, 32% 3%, 4% and 3%). Frequencies of wavy hair-grass *Deschampsia flexuosa,* common wood sorrel *Oxalis acetosella*, tufted hair-grass *D. cespitosa* and wood anemone *Anomone nemorosa* were similar (approximately 30%, 20%, 20% and 3% respectively). Relative cover of broad buckler-fern (46% vs 13%), common bracken (4% vs 19%), enchanter's-nightshade (15% vs 0%), Woodland germander (10% vs 3%), hairy wood-rush (8% vs 1%) and common wood sorrel (5% vs 2%) was higher in uncut forest than in clearcuts while relative cover the other species was similar. Forest herbaceous species were sampled using total of 82 quadrats (4 m^2) in four clearcut sites (0.5 – 2 ha) ages 6 – 13 years, and 219 grid cells (50 × 50 m) in uncut forest areas (surrounding the sites).

A replicated, randomized, controlled study in 2001-2002 in subtropical *Araucaria* forest in Brazil (9) found that **clearcutting and complete vegetation removal increased species richness and abundance of new seedlings**. Species richness/m^2 (clearcut: 0.7-2.0; uncut: 0.2-0.6) and abundance (clearcut: 0.8-2.6; uncut: 0.2-0.7) were higher in removal than uncut plots. Data were collected in 2002 in two removal (all plants and organic material removed in 2001) and two uncut plots (3 × 3 m) in each of 10 blocks randomly located inside a 2 ha area.

A replicated, controlled, before-and-after study in 1994-2000 in temperate broadleaf forest in Missouri, USA (10) found that **cutting increased plant species richness and ground vegetation cover**. Change in plant species richness was the greatest in clearcut, higher in group than single tree selection plots, and the lowest (and negative) in unthinned plots (clearcut: 21; group-selection: 12; single-tree-selection: 3; thinning: 7; unthinned: -3). Change in ground vegetation cover was the greatest in clearcut and the lowest in unthinned plots (clearcut: 43%; group selection: 17%; single tree selection: 10%; thinning: 9%; unthinned: 0%). Data were collected in 1994-1995 (before treatment) and in 1999-2000 (after treatment) in 45 group selection, 79 single tree selection and 36 thinning plots (25% trees reduction in all), and in 24 clearcut and 236 control plots (0.2 ha). Treatments were applied in 1996-1997.

A replicated, controlled study in 1998-2001 in boreal forest in Alberta, Canada (11) found that **clearcutting decreased tree density, basal area and cover as well as herbaceous cover and species richness**. Tree density (individuals/ha) (uncut: 568-1,069; thinned: 242-700; clearcutting: 26-52), basal area (m^2/ha) (uncut: 87-140; thinned: 39-105; clearcutting: 6-12) and canopy cover (uncut: 44-49%; thinned: 38-78%; clearcutting: 9-13%) were higher in uncut than clearcut plots. Herbaceous cover (uncut: 72-78%; thinned: 26-74%; clearcutting: 34-36%) and number of species/1 m^2 (uncut: 9-10; thinned: 7-9; clearcutting: 6) were higher in uncut than clearcut plots. Trees were monitored in 12 uncut, 18 thinned (20-75% retention) and nine clearcut (2% retention) plots (50 m^2). Herbaceous species were monitored in 1 × 1 m subplot within each plot. Treatments were applied in 1998-1999, monitoring was two years after treatment.

A replicated, controlled study in 1995-2005 in boreal forest in Ontario, Canada (12) found that **clearcutting increased the density and height of conifer regenerations and decreased the density of hardwood regenerations**. Conifer regeneration density (stems/ha) was higher in clearcut (8,000) than in partial-cut and uncut plots (2,000-3,000) Height was higher in clearcut and partial-cut plots (270-300 cm) than uncut plots (140 cm). Hardwood regeneration density was lower in clearcut (1,500) than partial-cut and uncut plots (4,500-5,000). Height was similar in all treatments (110-180 cm). Four plots (100 × 100 m): one uncut, two partial-cut and one clearcut (0%, 40% and 100% of merchantable trees basal-area removed) were replicated in four blocks in January-February 1995. Data were collected in 2005 in 12 subplots (2 × 2 m) within each plot.

A replicated, controlled, study in 2005-2007 in Mediterranean-type shrubland in Israel (13) found that **clearcutting increased herbaceous species richness under tree canopies**. Numbers of herbaceous species/0.1 ha plot was higher in clearcut (81) than in uncut plots (18). Ten uncut and 10 clearcut plots (0.1 ha) were established in December 2005. In April 2007, herbaceous species were monitored in uncut and in clearcut plots in areas that were covered with tree canopy before treatment.

A replicated, randomized, controlled study in 1998-2006 in boreal mixed wood forest in Alberta Canada (14) found that **clearcutting and high thinning intensity increased the cover of understory vegetation, particularly of grasses**. The average percentage cover of all species under the canopy was 67% in clearcut and high thinning intensity plots (0-10% retention) and 55% in the three low thinning intensities and uncut plots (20-100% retention). The average percentage cover of grasses was 14% in clearcut and high thinning intensity plots and 4% in the three low thinning intensities and uncut plots. Species richness was unaffected by retention level. In winter 1998-1999, six 10 ha compartments within each of three blocks (~2 km^2) were randomly assigned to six harvesting treatments: clearcutting (0% retention), 10% retention, 20% retention, 50% retention, 75% retention and uncut (100% retention). Sampling was conducted in summer 2006 in twenty 1×1 m plots in each of the 18 compartments.

A site comparison study in 2004 in tropical moist lowland forest in Hainan Island, China (15) found that **clearcutting increased species richness and abundance of lianas. Species richness** (species/ha) was 52 in the clearcut, 50 in the selective cut and 42 in the uncut site. Abundance of lianas (stems/ha) was 606 in the clearcut, 727 in the selective cut and 261 in the uncut site. Average diameter at breast height of liana stems was 2.7 cm in the clearcut, 3.1 cm in the selective cut and 4.5 cm in the uncut site. In 2004, a 1 ha plot was established in each of: uncut (since 1964) clearcut and selective cut (since 1964) forest sites. Each plot was divided to 100 quadrants (10 × 10 m). Liana stems were monitored in 2004.

A replicated, randomized, controlled study in 2003-2006 in temperate coniferous forest in Spain (16) found that **clearcutting reduced the number of maritime pine *Pinus pinaster* seedlings, and increased annual plant cover and the number of herbaceous plant species**. The number of **maritime pine** seedlings/plot was highest in uncut plots (66). It was higher in closed canopy (16) than in clearcut plots (1), and similar to both in open canopy plots (8). The number of annual species/plot was the highest in clearcut plots (41). It was higher in open canopy (31) than in uncut plots (24) and similar to both in closed canopy plots (29). The number of perennial herbaceous species/plot was highest in clearcut plots (17), and similar in open (11) and closed canopy (10) and uncut plots (9). Annual plant cover was lowest in uncut plots (21%), and similar in closed (37%) and open canopy (40%) and clearcut plots (42%). Perennial herbaceous species cover was higher in clearcut (16%) than open canopy plots (7%), and similar to both in uncut (11%) and closed canopy plots (10%). Three replicates (70 × 70 m) of each of four treatments: uncut; closed canopy (25% of basal area removed); open canopy (50% of basal area removed); clearcut (all trees removed) were established in 2003. Sampling was in 2006.

A replicated, controlled study in 2003-2007 in temperate coniferous forest in Germany (17) found that **clearcutting increased species richness and cover of understory vegetation**. Species richness (species/100 m^2) of herbaceous species (clearcut: 26-41; selection cutting: 26-32; control: 20-25) and cover of shrubs (clearcut: 14-21%; selection cutting: 13-20%; control: 0-3%) was similar between cutting treatments, but lower in control plots. Species richness of shrubs was higher in clearcut (8-9) than selection cutting plots (4-6) and lowest in control plots (2). Cover of herbaceous species was higher in clearcut (46-64%) than in control plots (18-48%) and similar to both those treatments in selection cutting plots (25-59%). In 2003, two plots (1 ha) of each treatment: clearcut (removing all trees), selection cutting (removing trees >45 cm DBH) and control (untreated) were established in each of two sites. Data were collected in 2007 in 20 subplots (100 m^2) in each plot.

A replicated, controlled study in 1993-2007 in boreal forest in Ontario, Canada (18) found **no effect of clearcutting on tree stem density**. Density was similar between treatments (clearcut: ~25,000 stems/ha; uncut: 23,560 stems/ha). Two replicates of clearcut (all merchantable timber removed) and uncut treatment sites were established in 1993. Data were collected in 2007 in ten 50 m^2 plots in each treatment site.

A replicated, randomized, controlled study in 1993-2007 in boreal forest in Ontario, Canada (19) found **no effect of clearcutting on the density of new tree regenerations.** Density (stems/ha) of hardwood and conifer regenerations was similar in all treatments (hardwood: ~4,000; conifer: ~1,500). In 1993-1994, four treatments: control (uncut), 50% partial cut, 50% partial cut with removal of residuals after 3 years, and clearcut were replicated in six blocks (112 × 56 m). Data were collected in 1998-2007.

A replicated, randomized, controlled study in 2000-2002 in boreal forest in Finland (20) found that **clearcutting decreased total plant biomass and the relative biomass of evergreen shrubs, but not deciduous shrubs, grasses or herbaceous species**. Total above ground biomass (clearcut: 12-20 g/m^2; control: 140-190 g/m^2) and relative biomass of evergreen shrubs (clearcut: 20-25%; control: 30-40%) were lower in clearcut. However, relative biomass of deciduous shrubs (53-67%) grasses (1-17%) and herbaceous species (1-7%) was similar between treatments. Data were collected in 2002 in 32 pairs of 0.5 × 0.5 m clearcut plots (biomass above the moss layer clipped each year 2002-2002) and 0.25 × 0.25 m control plots in each of two sites.

A replicated, paired sites study in 2003 in tropical forest in Brazil (21) found that **clearcutting decreased herbaceous density**. Herbaceous density (individuals/8 m^2 subplot) was lower in clearcut (3) than uncut (17) plots. Data were collected in 2003 in 25 subplots (4 × 2 m) within each plot (250 × 100 m) of three uncut (primary forest) and clearcut (secondary forest, cut in 1980-1984) pairs.

A before-and-after trial in 1992-2003 in boreal forest in Quebec, Canada (22) found that **clearcutting increased understory species richness, diversity and cover**. Numbers of species/1 m^2 plot increased in clearcut plots (before: 5-8; after: 9-13) and remained similar in uncut plots (before: 5-10; after: 6-12). Species diversity (Shannon's index) increased in clearcut plots (before: 0.7-1.2; after: 1.4-1.7) and remained similar in uncut plots (before: 0.7-1.3; after: 0.8-1.5). Cover increased in clearcut (before: 70-90%; after: 140-160%), but not uncut plots (before: 90-100%; after: 90-120%). In 1992, clearcut (all trees cut and removed) and uncut treatments (100 m^2) were replicated in three blocks (>625 m^2) at each of two sites. Data were collected before (1992) and after treatments (2003) in 5-12 plots (1 m^2) in each treatment.

A replicated, controlled study in 2006 in temperate woodland in south-eastern Australia (23) found **no effect of clearcutting on native plant species richness**. Numbers of native plant species/plot was similar between treatments (clearcut: 15; control: 19). Monitoring was in two pairs of clearcut and adjacent control plots (1 ha) in each of 12 sites (a total of 48 plots).

A before and after study in 1995-2003 in subtropical forest in Okinawa Island, Japan (24) found no effect of **clearcutting on tree species richness and diversity**. Species richness/plot (before: 33; after: 50) and Shannon's index of diversity (before: 4.0; after: 4.8) in cut plots were higher after treatment.

However, species richness (before: 24; after: 45) and diversity (before: 3.4; after: 4.5) were higher in uncut plots. Data were collected before (January 1995) and after treatment (2003) in five clearcut (in February 1995) and five uncut plots (10 × 10 m) established within a 4,000 m^2 forest section.

(1) Messina, M.G., Schoenholtz, S H., Lowe, M.W., Wang, Z., Gunter, D.K., and Londo, A.J. (1997) Initial responses of woody vegetation, water quality, and soils to harvesting intensity in a Texas bottomland hardwood ecosystem. *Forest Ecology and Management,* 90, 201-215.
(2) Jenkins, M.A., and Parker, G.R. (1998) Composition and diversity of woody vegetation in silvicultural openings of southern Indiana forests. *Forest Ecology and Management*, 109, 57-74.
(3) Jenkins, M.A., and Parker, G.R. (2000) The response of herbaceous-layer vegetation to anthropogenic disturbance in intermittent stream bottomland forests of southern Indiana, USA. *Plant Ecology*, 151, 223-237.
(4) Parrotta, J.A., Francis, J.K., and Knowles, O.H. (2002) Harvesting intensity affects forest structure and composition in an upland Amazonian forest. *Forest Ecology and Management,* 169, 243-255.
(5) Bourgeois, L., Messier, C., and Brais, S. (2004) Mountain maple and balsam fir early response to partial and clear-cut harvesting under aspen stands of northern Quebec. *Canadian Journal of Forest Research,* 34, 2049-2059.
(6) MacDonald, G.B., Cherry, M.L., and Thompson, D.J. (2004) Effect of harvest intensity on development of natural regeneration and shrubs in an Ontario boreal mixedwood stand. *Forest Ecology and Management,* 189, 207-222.
(7) Wienk, C.L., Sieg, C.H., and McPherson, G.R. (2004) Evaluating the role of cutting treatments, fire and soil seed banks in an experimental framework in ponderosa pine forests of the Black Hills, South Dakota. *Forest Ecology and Management,* 192, 375-393.
(8) Godefroid, S., Rucquoij, S. and Koedam, N. (2005) To what extent do forest herbs recover after clearcutting in beech forest? *Forest Ecology and Management,* 210, 39-53.
(9) Zanini, L., and Ganade, G. (2005) Restoration of araucaria forest: the role of perches, pioneer vegetation, and soil fertility. *Restoration Ecology,* 13, 507-514.
(10) Zenner, E.K., Kabrick, J.M., Jensen, R.G., Peck, J.E., and Grabner, J.K. (2006) Responses of ground flora to a gradient of harvest intensity in the Missouri Ozarks. *Forest Ecology and Management,* 222, 326-334.
(11) Macdonald, S.E., and Fenniak, T.E. (2007) Understory plant communities of boreal mixedwood forests in western Canada: Natural patterns and response to variable-retention harvesting. *Forest Ecology and Management,* 242, 34-48.
(12) Man, R., Kayahara, G.J., Rice, J.A., and MacDonald, G.B. (2008) Eleven-year responses of a boreal mixedwood stand to partial harvesting: Light, vegetation, and regeneration dynamics. *Forest Ecology and Management*, 255, 697-706.
(13) Agra, H., Ne'eman, G. (2009) Woody species as landscape modulators: their effect on the herbaceous plants in a Mediterranean maquis. *Plant Ecology,* 205, 165-177.
(14) Craig, A., and Macdonald, S.E. (2009) Threshold effects of variable retention harvesting on understory plant communities in the boreal mixedwood forest. *Forest Ecology and Management,* 258, 2619-2627.
(15) Ding, Y., and Zang, R. (2009) Effects of logging on the diversity of lianas in a lowland tropical rain forest in Hainan Island, South China. *Biotropica,* 41, 618-624.
(16) González-Alday, J., Martínez-Ruiz, C., and Bravo, F. (2009) Evaluating different harvest intensities over understory plant diversity and pine seedlings, in a *Pinus pinaster* Ait. natural stand of Spain. *Forest Ecology and Management,* 262, 404-412.
(17) Heinrichs, S., and Schmidt, W. (2009) Short-term effects of selection and clear cutting on the shrub and herb layer vegetation during the conversion of even-aged Norway spruce stands into mixed stands. *Forest Ecology and Management*, 258, 667-678.
(18) Li, J., Dang, Q.L, and Ambebe, T.F. (2009) Post-fire natural regeneration of young stands on clearcut and partial-cut and uncut sites of boreal mixedwoods. *Forest Ecology and Management,* 258, 256-262.
(19) Man, R., Rice, J.A., and MacDonald, G.B. (2009) Long-term response of planted conifers, natural regeneration, and vegetation to harvesting, scalping, and weeding on a boreal mixedwood site. *Forest Ecology and Management,* 258, 1225-1234.

(20) Manninen, O.H., Stark, S., Kytöviita, M.M., Lampinen, L., and Tolvanen, A. (2009) Understory plant and soil responses to disturbance and increased nitrogen in boreal forests. *Journal of Vegetation Science*, 20, 311-322.
(21) Ribeiro, M.B.N., Bruna, E.M., and Mantovani, W. (2010) Influence of Post-Clearing Treatment on the Recovery of Herbaceous Plant Communities in Amazonian Secondary Forests. *Restoration Ecology,* 18, 50-58.
(22) Grandpré, L., Boucher, D., Bergeron, Y., and Gagnon, D. (2011) Effects of small canopy gaps on boreal mixedwood understory vegetation dynamics. *Community Ecology*, 12, 67-77.
(23) Dorrough, J., Mcintyre, S., Brown, G., Stol, J., Barrett, G., and Brown, A. (2012) Differential responses of plants, reptiles and birds to grazing management, fertilizer and tree clearing. *Austral Ecology,* 37, 569-582.
(24) Wu, L., Deng, Y., Watanabe, S., Takakazu, S., Qin, J., Zhong, J., and Zhang, J. (2013) Early response of stand structure and species diversity to strip-clearcut in a subtropical evergreen broad-leaved forest in Okinawa Island, Japan. *Forest Ecology and Management,* 18, 414-417.

6.8. Use shelterwood harvesting

- Six of seven studies (including five replicated, controlled studies) in Australia[3], Iran[7], Nepal[1] and the USA[2,5,6,8] found that shelterwood harvesting increased **abundance**[7], **species richness**[2,5] and **diversity**[5,6] of understory plants, as well as the growth[1] and survival rate[8] of young trees. One study found shelterwood harvesting decreased plant species richness and abundance[3]. One study found no effect of shelterwood harvest on trees abundance[7].
- One replicated, controlled study in Canada[4] found no effect of shelterwood harvest on red **oak acorn production**.

Background
Shelterwood harvesting is a management technique designed to obtain even-aged forests. It involves harvesting trees in a series of partial cuts, with trees removed uniformly over the plot. This allows new seedlings to grow from the seeds of older trees. This can help to maintain distinctive forest species and increase forest structural diversity.

A controlled study in 1993-1995 in subtropical moist forest in Nepal (1) found that **shelterwood harvest treatments increased the growth rate of the dominant shala tree *Shorea robusta* seedlings**. Height growth was higher in clearcut than other treatment plots (unharvested: 85 cm; 75 trees remaining: 90 cm; 25 trees remaining: 99 cm; clearcut: 127 cm). Diameter growth was higher in clearcut (24 mm) than 25 (17 mm) and 75 trees remaining plots (16 mm), and the lowest in unharvested plots (6 mm)Numbers of **shala tree** seedlings was similar between treatments (unharvested: 89,292; 75 trees: 73,542; 25 trees: 91,125; clearcut: 81,000). Four treatment plots (1 ha) were established in 1993: unharvested, and 25 trees, 75 trees and clearcut (shelterwood harvest leaving 75, 25 and no trees in the plot, respectively). Growth rate of the 2,000 dominant **shala tree** seedlings in each plot was calculated for the second growing season after treatments. Numbers of seedlings was determined eight months after treatments.

A replicated, controlled study in 1977-1997 in temperate mixed coniferous forest in California USA (2) found that **shelterwood harvest increased understory species richness**. Numbers of species/1.13 ha was higher in shelterwood (80) than in unharvested plots (48). The study area was divided in sections of 8–80 ha that were assigned to shelterwood harvest (approximately 40 seed-trees/ha were left) and unharvested (since early 1990s) treatments. Understory vegetation was monitored in 30 m radius plots within the section treated annually since 1977.

A site comparison study in 2000 in a Mediterranean jarrah forest in Western Australia (3) found that **shelterwood harvest decreased plant species richness and abundance**. The number of native plant individuals/m^2 (shelterwood: 29; unharvested: 39) and individuals/30 m^2 (shelterwood: 869; unharvested: 1,172), and the number of native plant species/m^2 (shelterwood: 9.6; unharvested: 11.9) were lower in shelterwood than unharvested plots. The number of species/30 m^2 was similar between treatments (shelterwood: 55; unharvested: 57). Data were collected in five lines of 30 quadrats (1 × 1 m) in shelterwood (retaining basal area of 13 m^2/ha, applied in 1995) and unharvested treatments located in an 11,000 ha study area.

A replicated, controlled study in 1993-1997 in temperate mixed forest in Ontario, Canada (4) found **no effect of shelterwood harvest on the acorn crop of red oak *Quercus rubra***. Acorn production (unharvested: 29,577; shelterwood: 28,697/plot) and the percentage of acorns damaged by insects (shelterwood: 44%; unharvested: 47%) were similar between treatments. Five shelterwood (trees cut to 50% crown cover) and five unharvested plots (60 m × 60 m) were established in 1993-1994. Acorns were sampled using one trap (1 × 1 m) per 120 m^2 of crown cover (total of 86 traps) in August-November 1997.

A replicated, controlled study in 1994-2000 in mixed hardwood forest in North Carolina USA (5) found that **shelterwood harvest increased the density and the diversity of plants**. The density (individuals/ha) of trees (shelterwood: 1,009-1,094; uncut: 771), density of shrubs (shelterwood: 38,269-49,117; uncut: 21,789), number of shrub species/plot (shelterwood: 10; uncut: 4) and diversity (Shannon's index) of herbaceous plants (shelterwood: 2.2-2.4; uncut: 1.8) were higher in shelterwood harvest treatments. In 1994, eight sites (4.0-6.6 ha) were each assigned to one of three treatments: three sites of two shelterwood treatments (5 m^2/ha and 9 m^2/ha residual basal area), and two uncut sites. Monitoring was in 2000 in four plots (20 × 40 m) in each treatment site.

A replicated, controlled study in 2000-2001 in temperate broadleaf forest in Tennessee, USA (6) found that **shelterwood harvest increased herbaceous species diversity**. Diversity of herbaceous species (shelterwood: 4.2; unharvested: 3.1) was higher in shelterwood, while that of woody plants (shelterwood: 2.7-2.9; unharvested: 2.3-2.8) was similar between treatments. In July 2001, shelterwood (leaving high-quality stems, retaining 11.5 m^2/ha basal area) and unharvested treatments were applied each to two plots (0.8 ha) in each of four sites (total of 16 plots). Data were collected after treatment in summer 2001. Simpson's index was calculated for 3.6 m radius circular subplot in each plot.

A replicated, controlled study in 1986-2005 in Hyrcanian forest in Iran (7) found that **shelterwood harvest increased the abundance of some herbaceous species, but not of trees**. The frequency of four out of 17 herbaceous species was higher in shelterwood (16-27%) than unharvested plots (1-17%). The frequency of the other 13 species was similar between treatments (shelterwood: 1-20%; unharvested: 0-12%). Density (individuals/ha) of the dominant tree species oriental beech *Fagus orientalis* (shelterwood: 100; unharvested: 103), as well as of another six tree species (shelterwood: 0-8; unharvested: 0-9) was similar between treatments. Trees density was measured in 60 unharvested and 60 shelterwood (20-25% intensity in 1986 and 1991) treatment plots (100 m^2). Herbaceous species were monitored in 1 m^2 subplots within the plots. Data were collected in 2005.

A replicated, controlled study in 2001-2009 in temperate mixed oak forest in Pennsylvania USA (8) found that **shelterwood harvest treatments increased the survival of oak *Quercus spp.* seedlings**. The number of surviving black oak *Quercus velutina* and northern red oak *Q. rubra* seedlings/32 m^2 plot was highest in complete harvest and large-scale harvest plots (180-220), lower in preparatory cut plots (100-150) and the lowest in uncut plots (25-50). The number of surviving chestnut oak *Quercus montana* seedlings was higher in the three cutting treatments (120-180) than in uncut plots (25). The number of surviving white oak *Quercus alba* seedlings was the highest in complete harvest plots (200), lower in large-scale harvest plots(150) and the lowest in preparatory cut and uncut plots (10-50). Four treatments: uncut, preparatory cut (harvest of intermediate trees), large-scale harvest and complete harvest were replicated at each of five sites. Monitoring was in four 8 × 4 m plots in each treatment, each planted with 400 seedling of one of the four oak species.

(1) Rautiainen, O., and Suoheimo, J. (1997) Natural regeneration potential and early development of *Shorea robusta* Gaertn. f. forest after regeneration felling in the Bhabar-Terai zone in Nepal. *Forest Ecology and Management,* 92, 243-251.

(2) Battles, J.J., Shlisky, A.J., Barrett, R.H., Heald, R.C., and Allen-Diaz, B.H. (2001) The effects of forest management on plant species diversity in a Sierran conifer forest. *Forest Ecology and Management,* 146, 211-222.

(3) Burrows, N.D., Ward, B., and Cranfield, R. (2002) Short-term impacts of logging on understory vegetation in a jarrah forest. *Australian Forestry,* 65, 47-58.

(4) Bellocq, M.I., Jones, C., Dey, D.C., and Turgeon, J.J. (2005) Does the shelterwood method to regenerate oak forests affect acorn production and predation? *Forest Ecology and Management,* 205, 311-323.

(5) Elliott, K.J., and Knoepp, J.D. (2005) The effects of three regeneration harvest methods on plant diversity and soil characteristics in the southern Appalachians. *Forest Ecology and Management,* 211, 296-317.

(6) Jackson, S.W., Harper, C.A., Buckley, D.S., and Miller, B.F. (2006) Short-term effects of silvicultural treatments on microsite heterogeneity and plant diversity in mature Tennessee oak-hickory forests. *Northern Journal of Applied Forestry,* 23, 197-203.

(7) Pourmajidian, M.R., Jalilvand, H., Fallah, A., Hosseini, S.A., Parsakhoo, A., Vosoughian, A., and Rahmani, A. (2010) Effect of shelterwood cutting method on forest regeneration and stand structure in a Hyrcanian forest ecosystem. *Journal of Forestry Research,* 21, 265-272.

(8) Brose, P.H. (2011) A comparison of the effects of different shelterwood harvest methods on the survival and growth of acorn-origin oak seedlings. *Canadian Journal of Forest Research*, 41, 2359-2374.

6.9. Use group-selection harvesting

- Four of eight studies (including one replicated, controlled study) in Australia[2], Canada[6], Costa Rica[5] and the USA[1,3,4,7,8,9] found that group-selection harvesting increased **cover**[7] and **diversity**[4,6] of **understory plants** and the **density of young trees**[5]. Two studies found it decreased **understory species richness**[2] and **biomass**[8].Two studies found no effect on understory species richness[1,3] and diversity[3] and two found no effect of group-selection harvest on **tree density**[4] and **growth-rate**[9].

Background

Group selection thinning, i.e. thinning by removing trees in groups, leaves open gaps is used as a conservation management practice to increase forest structural diversity.

A replicated, controlled study in 1977-1997 in temperate mixed coniferous forest in California USA (1) found **no effect of group- or single-selection harvesting on understory plant species richness**. Numbers of species/1.13 ha in group (58) and single-tree selection harvest (52) was similar to unharvested plots (48). The study area was divided in sections of 8–80 ha that were assigned to the following treatments: group and single tree selection (approximately 11% of the section was harvested every 10 years in groups smaller than 0.6 ha and smaller than 0.1 ha respectively) and unharvested (since early 1990s). Understory vegetation was monitored in 30 m radius plots within each treatment annually from 1977.

A site comparison study in 2000 in a Mediterranean jarrah forest in Western Australia (2) found that **group selection harvesting decreased plant species richness and abundance**. The number of native plant individuals/m^2 (group selection: 31; uncut: 38) and individuals/30 m^2 (group-selection: 943; uncut: 1,138), as well as the number of native plant species/m^2 (group-selection: 10.1; uncut: 13.3) were lower in group-selection than uncut plots. The number of species/30 m^2 was similar between treatments (group-selection: 53; uncut: 57). Data were collected in five lines of 30 quadrats (1 × 1 m) in group-selection (retaining gaps of 4–7 ha, applied in 1995) and uncut treatments located in an 11,000 ha study area.

A replicated, controlled study in oak–pine *Quercus–Pinus* forest in Maine, USA (3) found **no effect of group selection harvesting on species richness and diversity of understory vegetation**. Numbers of species/1 m^2 (group-selection: 18-34; uncut: 18-25) and diversity (Shannon's index group-selection: 1.7-2.2; uncut: 1.9-2.1) were similar between treatments. Data were collected in 1998 in 40 pairs of group-selection (36-3,393 m^2 gaps harvested in 1987-1988) and uncut sites inside a 40 ha study area. Equal number (proportional to the gap size) of 1 m^2 plots were monitored in each pair.

A replicated, controlled study in 1994-2000 in mixed hardwood forest in North Carolina USA (4) found that **group-selection harvesting increased the diversity of shrubs and herbaceous plants, but not the density of shrubs and trees**. Numbers of shrub species/plot (group-selection: 10; uncut: 4) and diversity (Shannon’s index) of herbaceous plants (group-selection: 2.2; uncut: 1.8) were higher in group-selection than uncut plots. The density (individuals/ha) of shrubs (group-selection: 28,347; uncut: 21,789) and of trees (group-selection: 742; uncut: 771) was similar between treatments. Three group-selection (0.1–0.2 ha openings, 25% tree-cover removed) and two uncut sites (4.0-6.6 ha) were established in 1994. Monitoring was in 2000 in four plots (20 × 40 m) in each treatment site.

A replicated, controlled study in 1997-1999 in tropical forest in Costa Rica (5) found that **group-selection harvesting increased the density of new tree**

seedlings. The density of new tree seedlings was 2.5/m^2 in group-selection, and <0.5/ m^2 in uncut plots. In 1997, large gaps (320–540 m^2) were created inside five 40 × 40 m plots (group-selection) by cutting and removing all stems ≥5 cm diameter at breast height. Five other similar size plots (uncut) were unmanipulated with respect to canopy cover. Data were recorded every two months for one year after treatment.

A replicated, controlled, before-and-after trial in 2004-2005 in temperate broadleaf forest in Ontario Canada (6) found that **group-selection harvesting increased the diversity of early spring herbaceous species and decreased the percent of plant species lost**. The increase in the diversity (Shannon's index) of early spring herbaceous species was higher in group-selection (0.15 to 0.25) than in unharvested plots (0.32 to 0.34). Overall, the percentage of plant species lost was higher in unharvested (15%) compared to the group-selection treatment (8%). The percentage of plant species gained was similar (unharvested: 29%; group-selection: 35%). Two replicates (average 33 ha) of each group-selection harvest (creating five 400 m^2, four 700 m^2 and three 1,400 m^2 gaps) and unharvested plots, were established between November 2004 and April 2005. Sampling was in April 2004 (pre-harvesting) and in April-May 2005 (post-harvesting) in 4 m^2 regeneration growth plots (45×2 in control and 112×2 in group-selection).

A replicated, controlled, before-and-after study in 1995-2001 in temperate coniferous forest in Oregon, USA (7) found that **group-selection harvesting increased the change over time in herbaceous and shrub cover**. The increase in herbaceous cover (group-selection: 3; uncut: -2%) and in low shrub cover (group-selection: 20%, uncut: -4%) was higher in group-selection than in uncut plots. The increase in bryophyte cover (group-selection: 14%; uncut: 5%) and in tall shrub cover (group-selection: 6%; uncut: 0%) was similar between treatments. Gaps (0.2 ha circular gaps, retaining 250 trees/ha) and uncut treatment units (15-53 ha) were established in each of four sites in 1995-1997. In uncut units about 7.5% of the area was covered using 0.1 ha circular plots. In group-selection units, one 0.1 ha plot was placed in each of 10 gaps, 10 gap-edges, and 10 areas between the gaps. Data were collected before treatments and again in 2001 in 16 subplots of 0.1 m^2 in each plot.

A replicated, randomized, controlled study in 2004-2008 in temperate broadleaf forest in Wisconsin, USA (8) found that **group-selection harvesting decreased the above ground biomass, but not the annual biomass increase**. Above ground biomass was lower in group-selection (242,000 kg/ha) than in uncut plots (260,000 kg/ha), while the annual biomass increase was similar between treatments (11,000 kg/ha). Biomass of all plants <1.4 m tall was higher in large (700 kg/ha) than in medium (620 kg/ha) and small (480 kg/ha) gaps, and was higher in all gap-sizes compared with the transition zones (250-300 kg/ha). In 2007, all trees >5 cm diameter at breast height were cut in one small, one medium and one large circular subplots (gaps) of 4, 8 and 11 m radius in each of 15 plots of 80 × 80 m group-selection. In other 20 similar plots, subplots remained uncut. Each subplot was surrounded by an untreated transition zone 4, 8, and 11 m wide respectively. Total above ground biomass was determined for the entire plot, biomass of plants <1.4 m tall was measured in four 2 × 2 m quadrat at each gap and transition zones.

A replicated, controlled, before-and-after study in 1995-2007 in mixed conifer and broadleaf temperate forest in Maine, USA (9) found that **two group-selection harvesting treatments affected tree annual growth rates differently, but neither differed from the uncut control**. Average basal area annual growth was higher in the large group (0.27 m^2/ha) than small group treatment (−0.05 m^2/ha). There was no difference in average basal area annual growth between any of the group-selection treatments and the uncut treatment (−0.09 m^2/ha). Three treatments were replicated at three different sites: large-group (trees removed from 20% of the area creating 1,000-2,000 m^2 gaps); small-group (trees removed from 10% of the area creating 500-1,000 m^2 gaps); and uncut. Treatments were applied in 1995-1997. Monitoring was in 2005-2007 in 20 plots (0.05 ha) randomly selected in each treatment.

(1) Battles, J.J., Shlisky, A.J., Barrett, R.H., Heald, R.C., and Allen-Diaz, B.H. (2013) The effects of forest management on plant species diversity in a Sierran conifer forest. *Forest Ecology and Management,* 146, 211-222.

(2) Burrows, N.D., Ward, B., and Cranfield, R. (2002) Short-term impacts of logging on understory vegetation in a jarrah forest. *Australian Forestry,* 65, 47-58.

(3) Schumann, M.E., White, A.S., Witham, J.W. (2003) The effects of harvest-created gaps on plant species diversity, composition, and abundance in a Maine oak-pine forest. *Forest Ecology and Management,* 176, 543-561.

(4) Elliott, K.J., and Knoepp, J.D. (2005) The effects of three regeneration harvest methods on plant diversity and soil characteristics in the southern Appalachians. *Forest Ecology and Management,* 211, 296-317.

(5) Dupuy, J.M., and Chazdon, R.L. (2008) Interacting effects of canopy gap, understory vegetation and leaf litter on tree seedling recruitment and composition in tropical secondary forests. *Forest Ecology and Management,* 255, 3716-3725.

(6) Falk, K.J., Burke, D.M., Elliott, K.A., and Holmes, S.B. (2008) Effects of single-tree and group selection harvesting on the diversity and abundance of spring forest herbs in deciduous forests in southwestern Ontario. *Forest Ecology and Management,* 255, 2486-2494.

(7) Davis, L.R., and Puettmann, K.J. (2009) Initial Response of Understory Vegetation to Three Alternative Thinning Treatments. *Journal of Sustainable Forestry,* 28, 904–934.

(8) Dyer, J.H., Gower, S.T., Forrester, J.A., Lorimer, C.G., Mladenoff, D.J., and Burton, J.I. (2010) Effects of selective tree harvests on aboveground biomass and net primary productivity of a secondgrowth northern hardwood forest. *Canadian Journal of Forest Research*, 40, 2360-2369.

(9) Arseneault, J.E., Saunders, M.R., Seymour, R.S., and Wagner, R.G. (2011) First decadal response to treatment in a disturbance-based silviculture experiment in Maine. *Forest Ecology and Management,* 262, 404-412.

6.10. Use herbicide to thin trees

- One replicated, controlled study in Canada[1] found no effect of using herbicide to thin pine trees on **total plant species richness**.

Background

Although the use of herbicides is often not recommended as a conservation tool, in some cases it is used to increase forest structural diversity.

A replicated, controlled study in 1993-1998 in temperate lodgepole pine *Pinus contorta* forest in British Columbia, Canada (1) found **no effect of using herbicide to thin lodgepole pine on total tree density or on total plant species richness**. There was no effect of herbicide on the number of trees (herbicide: 4,180; control: 7,648/ha) or number of plants species (herbicide: 24;

control: 23/treatment unit). Data were collected in 1998 in herbicide (using glyphosate herbicide to retain 1,000 stems/ha) and control treatment units (2-13 ha). Units were established in 1993 in each of three study areas.

(1) Sullivan, T.P., Sullivan, D.S., Lindgren, P.M.F., and Boateng, J.O. (2002) Influence of conventional and chemical thinning on stand structure and diversity of plant and mammal communities in young lodgepole pine forest. *Forest Ecology and Management,* 170, 173-187.

6.11. Thin trees by girdling (cutting rings around tree trunks)

- One before-and-after trial in Canada[1] found that thinning trees by girdling (cutting rings around tree trunks) increased **understory plant species richness, diversity and cover**.

Background
Girdling, i.e. thinning trees by cutting rings around their trunk, is used as a conservation management practice to increase forest structural diversity. This method imitates the natural death of a tree without using chemicals or cutting trees.

A before-and-after trial in 1992-2003 in boreal forest in Quebec, Canada (1) found that **thinning trees by girdling increased understory species richness, diversity and cover**. The number of species increased following girdling (before: 5-9; after: 7-12/1 m^2 plot) and remained similar in uncut plots (before: 5-10; after: 6-12). Species diversity increased following girdling (Shannon's index before: 0.8-1.1; after: 1.2-1.7) and remained similar in uncut plots (before: 0.7-1.3; after: 0.8-1.5). Plant cover increased following girdling (before: 80%; after: 100-120%) and remained similar in uncut plots (before: 90-100%; after: 90-120%). In 1992, girdling (cutting >1 cm deep cuts around the trunks of all conifers) and uncut treatments (100 m^2) were replicated in three blocks (>625 m^2) at each of two sites. Data were collected before (1992) and after treatments (2003) in 5-12 plots (1 m^2) in each treatment.

(1) Grandpré, L., Boucher, D., Bergeron, Y., and Gagnon, D. (2011) Effects of small canopy gaps on boreal mixedwood understory vegetation dynamics. *Community Ecology*, 12, 67-77.

6.12. Use thinning followed by prescribed fire

- Three of six studies (including one replicated, randomized, controlled study) in the USA found that thinning followed by prescribed fire increased **cover**[5] and **abundance**[1] **of understory plants as well as the density of deciduous trees**[6]. One study found that thinning then fire decreased trees density and **species richness**[5]. Three studies found no effect or mixed effects of thinning followed by prescribed fire on **tree growth rate**[4] and density of young trees[3,7].
- One replicated, controlled study Australia[2] found no effect of thinning followed by prescribed fire on the **genetic diversity of black ash**.

Background
Mechanical thinning of trees followed by prescribed fire is used as a conservation management practice to encourage forest renewal and to increase forest structural diversity.

A replicated, controlled study in 1999 in temperate mixed forest in Arizona USA (1) found that **thinning followed by prescribed burning increased the abundance of native grasses and species richness of exotic herbaceous species**. The abundance index of native grass species was higher in thinned and burned (48) than in untreated plots (19). The number of species/375 m^2 of exotic herbaceous species was higher in thinned and burned (4) than in untreated plots (2). The abundance index of native herbaceous species (30 vs 26), exotic herbaceous species (6 vs 3) and exotic grass species (6 vs 0), and the number of species/375 m^2 of native herbaceous species (19 vs 18), native grasses (6 in both) and exotic grasses (1 vs 0) were similar between thinned and burned and untreated plots. Data were collected in ten 375 m^2 plots in each of four thinned and burned forest fragments (30% of basal area removed between 1987 and 1993 and burned between 3-4 years of thinning) and four untreated forest fragments (20-80 ha).

A replicated, controlled study in in temperate eucalyptus woodland in Victoria Australia (2) found **no effect of harvesting followed by burning on genetic diversity of black ash *Eucalyptus sieberi***. Genetic diversity did not differ between thinned and burned and untreated plots (allelic richness: 7.7 vs 8.0; effective number of alleles: 3.2 vs 3.4; expected heterozygosity:0.49 vs 0.50 respectively). Black ash seedlings were sampled in two 5 ha thinned and burned (cutting to retain ~10% of trees followed by prescribed burning in 1989-1990) and two 5-7 ha untreated plots. Molecular analysis was carried out using 35 Mendelian markers.

A replicated, controlled study in 2001-2005 in second-growth oak forests in southern Ohio, USA (3) found that **mechanical thinning followed by prescribed fire reduced large sapling density, increased small sapling and large seedling density, but did not affect densities of small seedlings and of oak *Quercus spp.* saplings**. Densities of large seedlings (50-150 cm tall) and small saplings (<3 cm DBH) was higher in thinned and burned (11,000 large seedlings/ha; 3,000 small saplings/ha) than in untreated plots (1,500 large seedlings/ha; 1,000 small saplings/ha). The density of large saplings (3-10 cm DBH) was lower in thinned and burned plots (200 large saplings/ha) than in untreated plots (600 large saplings/ha). The density of small seedlings (<50 cm tall) was similar in thinned and burned (90,000 small seedlings/ha) and in untreated plots (120,000 small seedlings/ha). Three forest areas were divided into treatment units (each approximately 30 ha): untreated, mechanical thinning followed by prescribed fire. Treatments were applied in the inactive season of 2001. Regeneration was sampled in ten 0.1 ha plots/treatment (a total of 40 plots/site) in summer 2004.

A replicated, controlled study in 2003-2005 in temperate coniferous forest in Montana USA (4) found **no effect of selection cutting followed by spring prescribed burning on tree growth rate**. Tree basal area increase overten years was not significantly different between thinned and burned (107 cm^2) and untreated plots (75 cm^2). One thinned and burned plot (selection cutting followed by spring prescribed burning in 1992-1993) and one untreated plot (50 × 50 to 60 × 60 m) were established at each of three sites. Trees were measured in 1992-1993 and again in 2003.

A replicated, randomized, controlled study in 2000-2003 in temperate mixed forest in Georgia, USA (5) found that **thinning followed by burning decreased tree density and species richness, and increased the cover of understory plants**. The number of trees/ha (winter: 215; spring: 305; summer: 258; untreated: 793) and the number of tree species/100 m^2 (winter: 4.3; spring: 6.0; summer: 3.3 untreated: 8.7) were lower in all thinned and burned treatments than in untreated plots. Understory plant cover (winter: 130%; spring: 113%; summer: 114%; untreated: 71%) was higher in thinned and burned treatments than in untreated plots. In 2000, three thinned and burned (mulching of all broadleaf trees regardless of size, and all pines <20 cm diameter at breast height followed by winter/spring/summer prescribed fire) and one unmanipulated treatment were randomly assigned to four plots (110 × 110 m) in each of four blocks. Data were collected in 2002-2003 in five subplots (10 × 10 m) within each treatment plot.

A replicated, controlled study in 2000-2007 in temperate conifer forest in Oregon USA (6) found that **mechanical thinning followed by prescribed burning increased the density of deciduous tree species**. The density of deciduous species (trees/ha) was higher in thinned and burned (84) than untreated plots (20). Two 1 ha plots were established in each of three thinned and burned sites (mechanical thinning followed by prescribed burning between 2000 and 2003) and three untreated sites. Data were collected from 2005 to 2007.

A before-and-after study in 2003-2005 in temperate coniferous forest in California, USA (7) found that **prescribed fire following thinning increased seedling density and decreased sapling density of conifers**. The change in density (after minus before individuals/ha) of seedlings <1.37 m tall (thinned and burned: -98; control: -2,303) was lower (more negative) in control plots. In contrast, the change in density of saplings >1.37 m tall and <10 cm diameter at breast height was lower in burned plots (burned: -740; control: 74). Data were collected in 2003 (before) and 2005 (after) in five plots (0.04 ha) in each of two thinned and burned (thinned to residual 30 m^2 basal area in June 2003 and burned in June 2004) and two control treatment units of approximately 1 ha each.

(1) Griffis, K.L., Crawford, J.A., Wagner, M.R., and Moir, W.H. (2001) Understory response to management treatments in northern Arizona ponderosa pine forests. *Forest Ecology and Management,* 146, 239-245.

(2) Glaubitz, J.C., Wu, H.X., Moran, G.F. (2003) Impacts of silviculture on genetic diversity in the native forest species Eucalyptus sieberi. *Conservation Genetics*, 4, 275-287.

(3) Albrecht, M.A., and McCarthy, B.C. (2006) Effects of prescribed fire and thinning on tree recruitment patterns in central hardwood forests. *Forest Ecology and Management,* 226, 88-103.

(4) Fajardo, A., Graham ,J.M., Goodburn, J.M., and Fiedler, C.E. (2007) Ten-year responses of ponderosa pine growth, vigor, and recruitment to restoration treatments in the Bitterroot Mountains, Montana, USA. *Forest Ecology and Management,* 243, 50-60.

(5) Brockway, D.G., Outcalt, K.W., Estes, B.L., and Rummer, R.B. (2009) Vegetation response to midstorey mulching and prescribed burning for wildfire hazard reduction and longleaf pine (*pinus palustris* mill.) ecosystem restoration. *Forestry,* 82, 299-314.

(6) Endress, B.A., Wisdom, M.J., Vavra, M., Parks, C.G., Dick, B.L., Naylor, B.J., and Boyd, J.M. (2012) Effects of ungulate herbivory on aspen, cottonwood, and willow development under forest fuels treatment regimes. *Forest Ecology and Management,* 276, 33-40.
(7) Walker, R.F., Fecko, R.M., Frederick, W.B., Johnson, D.W., and Miller, W.W. (2012) Seedling recruitment and sapling retention following thinning, chipping, and prescribed fire in mixed Sierra Nevada conifer. *Journal of Sustainable Forestry,* 31, 747-776.

6.13. Reintroduce large herbivores

- We found no evidence for the effects of reintroducing large herbivores on forests.

6.14. Pollard trees (top cutting or top pruning)

- We found no evidence for the effects of tree pollarding on forests.

Background
Pollarding is a pruning technique in which the upper branches of a tree are removed, promoting a dense head of foliage and branches.

6.15. Coppice trees

- We found no evidence for the effects of tree coppicing on forests.

Background
Coppicing is a traditional method of woodland management. Young tree stems are repeatedly cut down to ground level resulting in regrowth that can be harvested.

6.16. Halo ancient trees

- We found no evidence for the effects of haloing ancient trees on forests.

Background
As trees reach old age, they become smaller and their canopy becomes sparse because of the dieback of their outermost branches. As a result, ancient trees in dense forest stand the risk of being overtopped by younger, taller trees. Haloing involves the removal of these young, competing trees from around the ancient tree. This may release ancient trees from competition and allow them to survive for longer. However, sudden changes in environmental conditions (such as light availability) due to the removal of the surrounding canopy may also damage or kill ancient trees.

6.17. Adopt conservation grazing of woodland

- We found no evidence for the effects of adopting conservation grazing of woodland.

6.18. Retain fallen trees

- We found no evidence for the effects of retaining fallen trees on forests.

Background

Fallen trees may protect seedlings from large herbivores and hence stimulate natural regeneration. Furthermore, for some species, the dead wood may provide a substrate for seedlings.

6.19. Imitate natural disturbances by pushing over trees

- We found no evidence for the effects of imitating natural disturbances by pushing over trees on forests.

Background

Many forests in human-dominated landscapes are heavily managed reducing the likelihood of impacts from natural disturbances such as being blown over by strong winds or damage by large herbivores (which are often absent). Pushing or pulling over trees may imitate some of these mechanical disturbances and create a more heterogeneous environment, which may increase species diversity.

7. Threat: Invasive and other problematic species

Invasions by non-native species are considered a major threat on biodiversity (IUCN 2000). The impacts of invasive species are often severe and difficult to reverse. They can be as damaging to native species and ecosystems on a global scale as the loss and degradation of habitats.

IUCN (2000) *Guidelines for the prevention of biodiversity loss caused by alien invasive species*. Prepared by the SSC Invasive Species Specialist Group. Approved by the 51st Meeting of the IUCN Council, Gland Switzerland, February 2000.

Key messages – invasive plants

Manually/mechanically remove invasive plants

Two replicated, controlled studies in Hawaii and Ghana found that removing invasive grass or weed species increased understory plant biomass or tree seedling height. Two replicated, controlled studies in the USA and Hawaii found no effect of removing invasive shrubs or plants on understory plant diversity or growth rate of native species.

Use herbicides to remove invasive plant species

One replicated, randomized, controlled study in the USA found no effect of controlling invasive plants using herbicide on native plant species richness.

Use grazing to remove invasive plant species

We found no evidence of the effects of using grazing to remove invasive plant species on forests.

Use prescribed fire to remove invasive plant species

We found no evidence of the effects of using prescribed fire to remove invasive plant species on forests.

Key messages – native plants

Manually/mechanically remove native plants

We found no evidence of the effects of manually or mechanically removing native plants on forests.

Key messages - large herbivores

Use wire fences to exclude large native herbivores

Two replicated, controlled studies in the USA found that excluding large herbivores increased tree density. One of three studies, including two replicated, paired-sites, before-and-after studies, in Canada, Bhutan and Ireland found that excluding large herbivores increased the biomass of young trees. One found it decreased the density of young trees and one found mixed effects on species. Five of 10 studies, including two replicated, randomized, controlled studies, across the world found that excluding large herbivores increased the cover or and size of understory plants. Six found no effect on the cover, seed density, species richness or diversity of understory plants.

Use electric fencing to exclude large native herbivores
One controlled study in South Africa found that using electric fencing to exclude elephants and nyalas increased tree density.

Control large herbivore populations
We found no evidence of the effects of controlling large herbivore populations on forests.

Use fencing to enclose large herbivores (e.g. deer)
We found no evidence of the effects of using fencing to enclose large herbivores on forests.

Key messages - medium sized herbivores

Control medium-sized herbivores
We found no evidence of the effects of controlling medium-sized herbivores on forests.

Key messages - rodents

Control rodents
One controlled study in New Zealand found that rodent control decreased native plant species richness and had no effect on total plant species richness.

Key messages - birds

Control birds
One controlled study in Australia found that removing birds did not improve the health of the trees in a narrow-leaved peppermint forest.

Invasive plants

7.1. Mechanically/manually remove invasive plants

- One replicated, controlled study in Hawaii[1] found that removal of invasive grass species increased **understory plant biomass**. One replicated, controlled study in the USA[3] found no effect of invasive shrub removal on **understory plant diversity**.
- One replicated, controlled study in Ghana[2] found that removal of invasive weed species increased **tree seedling height**.
- One replicated, controlled study in Hawaii[4] found no effect of invasive plant removal on **growth rate of native species**.

Background
Invasions by non-native species are considered a major threat to biodiversity. One way of controlling invasive plants is by selective mechanical removal. This

action may also indirectly affect the abundance of plants other than the targeted species, and can as result influence the forest structure and composition.

A replicated, controlled study in 1991-1994 in dry tropical forest in Hawaii, USA (1) found that **removal of invasive grass species increased the relative growth rate and biomass of two of four native shrubs**. For hopbrush *Dodonaea viscosa* and Hawaiian hawthorn *Osteomeles anthyllidifolia* changes in basal circumference (removal: 13% and 13%; control: 7% and 5%, respectively) and biomass (removal: 33% and 40%; control: 12% and 13%, respectively) were higher in removal than control plots. For maiele *Styphelia tameiameia* and *Metrosideros polymorpha* changes in basal circumference (removal: 5% and 9%; control: 4% and 9%, respectively) and biomass (removal: 10% and 3%; control: 6% and 2%, respectively) were similar between treatments. Trees of the four dominant shrub species were monitored in three control (untreated) and four removal (all introduced grasses removed) plots (20 × 20 m) established in spring 1991. Data were collected in 1992 and 1994.

A replicated, controlled study in 1998 in dry semi-deciduous forest in Ghana (2) found that **clearance of invasive Siam weed *Chromolaena odorata* increased the seedling height and number of leaves for 23 out of 25 native tree species**. For 25 out of 28 tree species height increases (removed: 2-14 cm; control 1-3 cm) and numbers of leaves/individual (removed: 1-8; control: 1-3) were higher in removal plots. In contrast, for the other three species, increases in height (2-4 cm) and numbers of leaves (1-2 leaves/individual) were similar between treatments. In May-June 1998, removal (Siam weed and all other non-tree plants removed) and control (no plant removal) treatments were applied to 54 circular plots each (1.3 m radius). A second weed removal was carried out in July 1998. Increases in height and numbers of leaves for tree seedlings ≤2 m tall were calculated from the difference between measurements taken in June (after the first removal) and September 1998.

A replicated, controlled study in 2001-2004 in temperate broadleaf forest in Ohio, USA (3) found **no effect of removal of the invasive shrub** Amur honeysuckle ***Lonicera maackii* after hydrologic restoration on understory plant species richness and diversity**. Numbers of species (cleared: 4; control: 4/m^2) and diversity (Shannon's index cleared: 0.8; control: 0.9) were similar between treatments. Data were collected in 2004 in 29 cleared (removal of Amur honeysuckle shrubs during 2001–2003 and herbicide treatment to the remaining stumps) and 31 control (uncleared) plots (1 m^2). A hydrologic restoration project had been carried out at the site in 2000-2001.

A replicated, controlled study in 2004-2007 in lowland wet forest in Hawaii, USA (4) found that **removal of all introduced species decreased leaf density but did not affect growth rate of native species**. The forest leaf density (leaf area index) was lower in removal (2.5 m^2 leaf/m^2) than in control plots (6 m^2 leaf/m^2). For the three main native species, relative growth rate (*Diospyros sandwicensis*: 0.1%; *Metrosideros polymorpha*: 0.2%-0.3%; *Psychotria hawaiiensis*: 3.5%-3.9%) and absolute diameter at breast height growth (*D. sandwicensis*: 0.01-0.02; *M. polymorpha*: 0.07; *P. hawaiiensis*: 0.16-0.19 cm/yr) were similar. In April–June 2004, four pairs of control and removal (all introduced species removed) treatment plots (10 × 10 m) were replicated in three transects. The forest leaf area index was measured in February 2006. Growth of native trees was evaluated in 2007.

(1) D'Antonio, C.M., Hughes, R.F., Mack, M., Hitchcock, D., and Vitousek, P.M. (2010) The response of native species to removal of invasive exotic grasses in a seasonally dry Hawaiian woodland. *Journal of Vegetation Science,* 9, 699-712.

(2) Honu, Y.A.K., Dang, Q.L. (2000) Responses of tree seedlings to the removal of *Chromolaena odorata* Linn. in a degraded forest in Ghana. *Forest Ecology and Management,* 137, 75-82.

(3) Swab, R.M., Zang, L., and Mitsch, W.J. (2008) Effect of hydrologic restoration and *Lonicera maackii* removal on herbaceous understory vegetation in a bottomland hardwood forest. *Restoration Ecology,* 16, 453-463.

(4) Ostertag, R., Cordell, S., Michaud, J., Cole, T.C., Schulten, J.R., Publico, K.M., and Enoka, J.H. (2009) Ecosystem and restoration consequences of invasive woody species removal in Hawaiian lowland wet forest. *Ecosystems*, 12, 503-512.

7.2. Use herbicides to remove invasive plant species

- One replicated, randomized, controlled study in the USA[1] found no effect of invasive plant control using herbicide on the total **native plant species richness**.

Background

Invasions by non-native species are considered a major threat to biodiversity. One way of controlling invasive plants is by using herbicides. This action may also affect the abundance of plants other than the targeted species, and as result influence the forest structure and composition.

A replicated, randomized, controlled study in 2000-2005 in temperate broadleaf forest in Ohio, USA (1) found **no effect of control of invasive garlic mustard *Alliaria petiolata* on native plant species richness and diversity**. Species richness was similar between treatments in both old-growth (sprayed: 8.7; unsprayed: 8.0 species/plot) and second-growth forests (sprayed: 10.6; unsprayed: 10.5). The same was true for species diversity (Shannon's index old-growth: sprayed 1.7, unsprayed 1.4; second-growth: sprayed 2.0, unsprayed 2.0). Data were collected in 2005 in 25 sprayed (garlic mustard individuals sprayed with glyphosate herbicide Roundup© PRO at the start of winter in 2000-2004) and 25 unsprayed plots (1×1 m). Plots were randomly placed in each of 16 ha second-growth and 20 ha old-growth forest sections.

(1) Hochstedler, W.W., Slaughter, B.S., Gorchov, D.L., Saunders, L.P., and Stevens, M.H.H. (2007) Forest floor plant community response to experimental control of the invasive biennial, *Alliaria petiolata* (garlic mustard). Journal of the Torrey Botanical Society, 134, 155-165.

7.3. Use grazing to remove invasive plant species

- We found no evidence for the effects of using grazing to remove invasive plant species on forests.

7.4. Use prescribed fire to remove invasive plant species

- We found no evidence for the effects of using prescribed fire to remove invasive plant species on forests.

Native plants

7.5. Manually/mechanically remove native plants

- We found no evidence for the effects of manually or mechanically removing native plants on forests.

Large herbivores

7.6. Use wire fencing to exclude large native herbivores

- Five of 10 studies (including two replicated, randomized, controlled studies) in Australia[1,11], Bhutan[5], Canada[6], France[8], Portugal[10] and the USA[2,3,7,9] found that using wire fencing to exclude large herbivores increased the **cover**[7,10] and **size**[5,6,9] of understory plants. Six studies found no effect of wire fencing on the cover[2], **seed density**[1], **species richness**[2,3,8,10,11] and **diversity**[2,3,8,] of **understory plants**.
- Two of the above studies[5,6] and one paired-sites study in Ireland[4] examined the effect of using wire fencing to exclude large herbivores on young trees. One found it increased the **biomass**[6], one found it decreased the **density**[5] **of young trees** and one found mixed effects[4] depending on the species.
- Two replicated, controlled studies in the USA found that using wire fencing to exclude large herbivores increased tree density[12,13].

Background

High grazing pressure by large herbivore can result in degraded understory species diversity, mainly due to decrease in the abundance of palatable herbaceous species. Excluding large herbivores from forests by creating exclosures using wire fences can increase species diversity.

A replicated, controlled study in 1997 in eucalypt woodlands in New South Wales, Australia (1) found **no effect of excluding large herbivores on topsoil seed density**. The density of topsoil seeds was similar between grazed and ungrazed plots (12,360 and 9,351 seeds/m^2 respectively). Topsoil (0-10 cm depth) seeds were monitored in five grazed (1-2 dry sheep equivalents/ha/year) and five ungrazed 20 ×20 m plots. Soil was sampled in 1997 using a soil corer 20 cm in diameter.

A replicated, randomized, controlled, study in 1977-1981 in a subtropical moist lowland forest in Alabama, USA (2) found that **excluding deer and cattle**

had no effect on plant cover, species richness or diversity after four growing seasons. Plant cover were 135, 132 and 138, numbers of species were 29, 29, 29 and species diversities (Shannon's index) were 2.37, 2.41 and 2.46 for ungrazed, deer-grazed and cattle-and-deer-grazed treatments respectively. Three 900 ha allotments, each containing six 150 ha blocks were established in 1977. Three treatments were randomly assigned to three 0.1 ha plots within each block: grazing by deer and cattle excluded, grazing by deer only and grazing by deer and cattle. Plant data were collected in September and October 1978–1981 along three 20 m line transects within each treatment plot.

A replicated, randomized, controlled study in 1996-2004 in temperate mixed forest in Tennessee, USA (3) found **no effect of excluding deer on spring flower species richness and diversity**. Numbers of species (exclosure: 1.5-6; unfenced: 2-6/100 m^2) and species diversity (Shannon's index exclosure: 0.25-0.75; unfenced: 0.25-0.90) were similar between treatments. Data were collected in 2004 in five exclosure (fenced to exclude deer browsing in 1996) and five control (unfenced) plots (10 × 10 m) in each of three sites.

A replicated, paired sites study in 1969-2001 in temperate broadleaf forest in Ireland (4) found that **excluding deer decreased the number of seedlings but increased the number of saplings and the height of common holly *Ilex aquifolium* and rowan *Sorbus aucuparia***. In yew *Taxus baccata* wood sites, the density of holly seedlings was lower in fenced plots (fenced: 0.4; unfenced: 2.1/m^2), whereas the density of rowan seedlings was similar between treatments (fenced: 0.2; unfenced: 0.2) Sapling density of both holly (fenced 0.7, unfenced <0.1) and rowan (fenced 0.4, unfenced 0.0, respectively) and juvenile height () (holly: fenced 45, unfenced 8cm; rowan***:*** fenced 70, unfenced 10 cm) was higher in fenced plots. In oak-wood sites, seedling density for both holly (fenced: 0.5; unfenced: 21.9) and rowan (fenced: <0.1; unfenced: 0.8) was lower in fenced plots. Sapling density for holly was higher in fenced plots (fenced: 3.0; unfenced: 0.5) and for rowan it was similar between treatments (fenced: 0.3; unfenced: <0.1). Sapling juvenile height was higher in fenced plots for both holly (fenced: 130; unfenced: 10) and rowan (fenced: 240; unfenced: 10). Data were collected in 2001 in three fenced plots in yew wood-type sites (764-1,036 m^2 deer-proof exclosures established in 1969-1970), four fenced plots in oak wood-type sites (225-1,090 m^2, established in 1974-1975) and seven adjacent unfenced plots (225-600 m^2).

A replicated, paired-sites, before-and-after study in 1997-2005 temperate mixed conifer forest in the Bhutan Himalayas (5) found that **excluding large herbivores increased bamboo *Yushania microphylla* growth but decreased seedling density of all conifer trees, particularly Himalayan hemlock *Tsuga dumosa* and Sikkim spruce *Picea spinulosa***. Eight years after treatment, the percentage cover of bamboo increased by 42% in grazed and 58% in ungrazed plots. The number of all conifer tree seedlings increased by 16,333/ha in grazed and only 166/ha in ungrazed plots. The number of Himalayan hemlock seedlings increased by 14,417/ha in grazed and decreased by 167/ha in ungrazed plots. The number of Sikkim spruce seedling increased by 667/ha in grazed and decreased by 166/ha in ungrazed plots. In 1996, five pairs of 4×6 m treatment plots: grazed (unfenced) and ungrazed (fenced to keep out large herbivores) were established in each of two sites. Each was divided into six 2×2 m subplots that were sampled repeatedly in 1997 at the time of treatment and again in 2005.

A replicated, paired-sites, before-and-after trial study in 1998-2006 in temperate broadleaf forest in Quebec, Canada (6) found that **excluding deer increased the above ground biomass of spring-flowering herbaceous species, small seedlings and large shrubs and trees, but not of summer-flowering herbaceous species, grasses, ferns and small deciduous shrubs**. Eight years after treatments, the above ground biomass of small and large spring-flowering herbaceous species had increased by 119% and -19% in grazed plots and 570% and 89% in ungrazed plots respectively. The biomass of small deciduous seedlings had decreased by 63% in grazed and 18% in ungrazed plots. The biomass of large deciduous shrubs and trees had increased by 99% in grazed and 418% in ungrazed plots. Excluding deer did not affect above ground biomass of summer-flowering herbaceous species, grasses, ferns and small deciduous shrubs. Six sites of two 625 m^2 treatment plots: grazed (control) and ungrazed (deer exclosure) were established in 1998. Above ground biomass (g/m^2) was estimated in 1998 and 2006 in twenty 2 × 0.1 m subplots in each plot.

A replicated, controlled study in 2002-2008 in temperate broadleaf forest in Pennsylvania, USA (7) found that **excluding herbivores increased fruit production and the cover of some under-canopy species**. Six years after treatment the total number of fruit/plot (fenced: 20-430; unfenced: 0-1), relative cover of the palatable herbaceous species painted trillium *Trillium undulatum*, sessile bellwort *Uvularia sessilifolia* and Indian cucumber-root *Medeola virginiana* (fenced: 0-3%; unfenced: <1%), cover of bramble *Rubus spp.* (fenced: 1%-25%; unfenced: <1%) and the number of tree saplings (fenced: 0-2; unfenced: <1.0/m^2) were higher in fenced than unfenced plots. The cover of hay-scented fern *Dennstaedtia punctilobula* was similar between treatments (0-70%). Data were collected in 2008 in three blocks of 12 fenced (2 m tall fence with 10 × 10 cm openings) and 12 unfenced plots (50 × 80 m). Plots were established in 2002 in an area subjected to high and constant deer herbivory pressure.

A replicated, controlled study in 2005-2008 in temperate forest in France (8) found **no effect of excluding deer browsing on species richness and diversity of trees and herbaceous species.** The number of woody plant species (deer exclusion: 8-10; unfenced: 7-10/m^2) and their species diversity (Shannon's index deer exclusion: 2.1-2.5; un-fenced: 1.9-2.1) and the number of herbaceous species (exclusion: 17-20; un-fenced: 13-17/m^2) and their species diversity (Shannon's index deer exclusion: 3.4-3.5; unfenced: 2.9-3.1) were similar between treatments. Data were collected in May 2008. At one site there were 60 sampling plots (1 m^2) inside a 1 ha fenced area (deer exclusion) and 60 similar plots inside a 1 ha open area (unfenced). At a second site there were 42 sampling plots (1 m^2) inside a 1.5 ha fenced area (deer exclusion) and 42 similar plots inside a 1.5 ha open area (unfenced). Both sites were regularly grazed by roe deer *Capreolus capreolus* and red deer *Cervus elaphus*. Exclosures were set up in March 2005.

A replicated, controlled study in 2001-2006 in Mediterranean-type shrubland in California, USA (9) found that **excluding deer increased shrub height. Shrub height was higher in deer exclusion** (68 cm) **than in unfenced plots** (55 cm). Five unfenced control and five deer exclusion (1.5 m fence constructed in 2001-2003) plots (2.5 m^2) were replicated in twenty areas (2 ha). Data were collected three years after treatment.

A replicated, paired-sites study in 1979-1990 in Mediterranean oak woodland in south-east Portugal (10) found that **excluding red deer *Cervus elaphus* and fallow deer *Dama dama* increased the biomass of herbaceous species and the relative cover of legumes *Fabaceae*, but did not affect the number of plant species**. The biomass of herbaceous species was 177 g/m^2 in ungrazed and 100 g/m^2 in grazed plots. Relative cover of legumes was 10% in ungrazed and 5% in grazed plots. The total number of plant species was similar in grazed (44) and ungrazed (42) plots. Five blocks of paired ungrazed (fenced) and grazed (unfenced, grazed mainly by red deer and fallow deer) plots (25×25 m) were established in the study area in 2001. In 2003, plant biomass and the relative cover of plants were measured in four subplots (2×4 m) within each plot.

A replicated, controlled study in 1981-2010 in Mulga *Acacia aneura* dry forest in Queensland, Australia (11) **found no effect of excluding herbivores on the number of plant species.** There was no difference between treatments for species richness of all plants (exclusion: 15; unfenced: 16 species/plot), annual grasses (exclusion: 2; unfenced: 3), perennial grasses (exclusion: 3; unfenced: 3), annual herbaceous species (exclusion: 5; unfenced: 5) and perennial herbaceous species (exclusion: 4; unfenced: 3).. In 1981-1983, two treatments (50 × 50 m plots) were replicated at three sites: control (unfenced) and fences to exclude all mammalian herbivores >200 g. Plant species richness was determined in 2008 in twenty 2 × 7 m subplots in each treatment.

A replicated, controlled study in 2000-2007 in temperate conifer forest in Oregon, USA (12) found that **excluding grazing herbivores increased the density of tree species**. The combined density of *Populus spp.* and willows *Salix spp.* was higher in herbivore exclusion (212 trees/ha) than in unfenced plots (66). The density of the most common species, cottonwood *P. trichocarpa* was 122 trees/ha in herbivore exclusion and 24 trees/ha in unfenced plots. Two 1 ha plots, one in an area with grazing by cattle *Bos taurus*, elk *Cervus elaphus*, and mule deer *Odocoileus hemionus* and one fenced herbivore -exclusion area were established in each of six sites. Data were collected from 2005 to 2007.

A replicated, controlled study in 1987-2008 in boreal forest in Minnesota, USA (13) found that **excluding deer and snowshoe hares *Lepus americanus* increased tree density, basal area and biomass**. Increases were higher in exclusion plots for tree density (unfenced: 81%, 1,617 to 3,219 /ha; exclusion: 274%, 1,375 to 4,836 /ha), basal area (unfenced: 50%, 15 to 23 m^2/ha; exclusion: 125%, 11 to 25 m^2/ha) and biomass (unfenced: 37%, 72 to 98 tons/ha; exclusion: 95%, 53 to 104 tons/ha). Data were collected in 1991 and 2008 in three exclusion (fenced to exclude deer and snowshoe hares in 1987-1990) and three control (unfenced) plots (0.25/ha).

(1) Grant, C.D. and MacGregor, C.M. (2001) Topsoil seed banks in grazed and ungrazed eucalypt woodlands at Newholme, Armidale, New South Wales, Australia. *New Zealand Journal of Botany,* 39, 471-481.

(2) Brockway, D.G., and Lewis, C.E. (2003) Influence of deer, cattle grazing and timber harvest on plant species diversity in a longleaf pine bluestem ecosystem. *Forest Ecology and Management,* 175, 49-69.

(3) Webster, C.R., Jenkins, M.A., and Rock, J.H. (2005) Long-term response of spring flora to chronic herbivory and deer exclusion in Great Smoky Mountains National Park, USA. *Biological Conservation,* 125, 297-307.
(4) Perrin, P.M., Kelly, D.L., and Mitchell, F.J.G. (2006) Long-term deer exclusion in yew-wood and oakwood habitats in southwest Ireland: Natural regeneration and stand dynamics. *Forest Ecology and Management,* 236, 356-367.
(5) Darabant, A., Rai, P.B., Tenzin, K., Roder, W., and Gratzer, G. (2007) Cattle grazing facilitates tree regeneration in a conifer forest with palatable bamboo understory. *Forest Ecology and Management,* 252, 73-83.
(6) Collard, A., Lapointe, L., Ouellet, J. P, Crête, M., Lussier, A., Daigle, C., and Côté, S.D. (2010) Slow responses of understory plants of maple-dominated forests to white-tailed deer experimental exclusion. *Forest Ecology and Management,* 260, 649-662.
(7) Huebner, C.D., Gottschalk, K.W., Miller, G.W., and Brose, P.H. (2010) Restoration of three forest herbs in the Liliaceae family by manipulating deer herbivory and overstorey and understory vegetation. *Plant Ecology and Diversity*, 3, 259-272.
(8) Pellerin, M., Saïd, S., Richard, E., Hamann, J-L., Dubois-Coli, C., and Hum, P. (2010) Impact of deer on temperate forest vegetation and woody debris as protection of forest regeneration against browsing. *Forest Ecology and Management,* 260, 429-437.
(9) Potts, J.B., Marino, E., and Stephens, S.L. (2010) Chaparral shrub recovery after fuel reduction: A comparison of prescribed fire and mastication techniques. *Plant Ecology,* 210, 303-315.
(10) Bugalho, M.N., Lecomte, X., Gonçalves, M., Caldeira, M.C., and Branco, M. (2011) Establishing grazing and grazing-excluded patches increases plant and invertebrate diversity in a Mediterranean oak woodland. *Forest Ecology and Management,* 261, 2133-2139.
(11) Fensham, R.J., Silcock, J.L., and Dwyer, J.M. (2011) Plant species richness responses to grazing protection and degradation history in a low productivity landscape. *Journal of Vegetation Science,* 22, 997-1008.
(12) Endress, B.A., Wisdom, M.J., Vavra, M., Parks, C.G., Dick, B.L., Naylor, B.J., and Boyd, J.M. (2012) Effects of ungulate herbivory on aspen, cottonwood, and willow development under forest fuels treatment regimes. *Forest Ecology and Management,* 276, 33-40.
(13) White, M.A. (2012) Stand structure and composition 32 years after precommercial thinning treatments in a mixed northern conifer stand in central Maine. *Forest Ecology and Management,* 269, 222-228.

7.7. Use electric fencing to exclude large native herbivores

- One controlled study in South Africa[1] found that using electric fencing to exclude elephants and nyalas increased **tree density**.

Background
Activity of large herbivores can result in physical damage and degraded understory species diversity. Excluding large herbivores from forest areas by creating exclosures using electric fences can increase species diversity.

A controlled study in 2005-2007 in Sand Forest in South Africa (1) found that **exclusion of elephant *Loxodonta africana* and nyala *Tragelaphus angasii* increased tree density**. The density of all trees was higher when both species were excluded than unfenced plots (unfenced: ~8,000/ha; elephant excluded: ~10,000; nyala and elephant excluded: ~14,000). The density of seedlings was higher when both species were excluded than unfenced plots (unfenced: ~5,000; elephant excluded: ~6,000; nyala and elephant excluded: ~8,500). There were no differences between treatments for the density of saplings (unfenced: ~2,000; elephant excluded: ~2,500; nyala and elephant excluded: ~3,200) and grown trees (unfenced: ~1,000; elephant excluded: ~1,500; nyala and elephant excluded: ~2,300). Data were collected in 2007 in 12 plots (20 ×20 m) of each treatment: unfenced (accessible to elephants and nyalas), elephant excluded (inside elephant-excluded area of 3.1 km^2 surrounded by electrified-wire) and nyala and elephant excluded (wire-fence exclosures to exclude nyalas inside the elephant-free area) treatments. Treatments were applied in 2005 in a 5.2 km^2 Sand Forest patch.

(1) Lagendijk, G., Mackey, R.L., Page, B.R., and Slotow, R. (2011) The Effects of Herbivory by a Mega- and Mesoherbivore on Tree Recruitment in Sand Forest, South Africa. *Plos One*, 6, e17983.

7.8. Control large herbivore populations

- We found no evidence of the effects of controlling large herbivore populations on forests.

7.9. Use fencing to enclose large herbivores (e.g. deer)

- We found no evidence of the effects of using fencing to enclose large herbivores on forests.

Medium sized herbivores

7.10. Control medium-sized herbivores

- We found no evidence of the effects of controlling medium-sized herbivores on forests.

Rodents

7.11. Control rodents

- One controlled study in New Zealand[1] found that rodent control decreased **native plant species richness** and did not affect **total plant species richness**.

Background
Many rodents feed on seeds. Others that feed on tree bark may cause tree death by girdling (damaging round tree trunks). Controlling rodent populations can minimize seed predation or girdling and thus increase the abundance or reduce death of some plant species.

A controlled study in temperate mixed forest in New Zealand (1) found that **rodent control decreased native plant species richness, but did not affect total plant species richness**. The number of native plant species/plot was lower in rodent control plots (33) than untreated plots (38). The numbers of non-native plant species/plot (untreated: 4; rodent control: 3) and total vascular plant species/plot (untreated: 40; rodent control: 37) were similar between treatments. Plants were monitored in 400 m^2 plots in each of 14 untreated and 27 rodent control forest fragments. Control was carried out using trap stations, largely for ship rats *Rattus rattus* and house mice *Mus musculus*.

(1) Burns, B.R., Floyd, C.G., Smale, M.C., and Arnold, G.C. (2011) Effects of forest fragment management on vegetation condition and maintenance of canopy composition in a New Zealand pastoral landscape. *Austral Ecology,* 36, 153-166.

Birds

7.12. Control birds

- One controlled study in Australia[1] found that removing bell-miners from narrow-leaved peppermint forests did not improve the health of the trees in the forest.

Background

Birds can consume seeds or physically damage trees. However, insectivorous birds may also control invertebrate herbivore populations. Some territorial species (such as bell-miners) may also displace other insectivorous birds and hence affect the impact of invertebrates on forests.

A controlled study in 1992–1995 in three sites in narrow-leaved peppermint *Eucalyptus radiata* forest in south eastern Victoria, Australia (1) found that **the removal of bell miners *Manorina melanophrys* did not improve tree health.** The change in tree health (an index based on crown size, crown density, the presence of dead branches and the shoot growth) did not differ between the plots where bell miners had been removed (-0.6), were present (-2.3) and a control plot where no bell miners occurred (-0.7). In June 1993, a total of 189 bell miners were removed from the experimental site and the surrounding area (2.7 ha), by mist netting and culling. The tree health index was based on the visual assessment of the health of 10 trees at each plot (50 x 50 m), following a standardized protocol.

(1) Clarke M. F. & Schedvin N. (1999) Removal of bell miners *Manorina melanophrys* from Eucalyptus radiata forest and its effect on avian diversity, psyllids and tree health. *Biological Conservation,* 88, 111–120.

8. Pollution

Key messages

Maintain/create buffer zones

One site comparison study in Australia found that a forest edge protected by a planted buffer strip had higher canopy cover and lower stem density, but similar understory species richness to an unbuffered forest edge.

Remove nitrogen and phosphorus using harvested products

We found no evidence for the effects of removing nitrogen and phosphorus using harvested products on forests.

8.1. Maintain/create buffer zones

- One site comparison study in Australia[1] found that a forest edge protected by a planted buffer strip had higher canopy cover and lower stem density, but similar understory species richness to an unbuffered forest edge.

Background

Buffers can be created to exclude undesirable outside influences from forest sections.

A site comparison in 2008 in two remnants of complex mesophyll vine forests in North Queensland, Australia (1) found that a forest edge protected by a planted buffer strip had higher canopy cover and lower stem density, but similar understory species richness to a forest edge with no buffer. Canopy cover in the buffered forest edge (approx. 90%) was higher than that along the edge with no planted buffer (approx. 75%). Similarly, stem density along the buffered edge (approx. 4 trees/m^2) was lower than along the unbuffered edge (approx. 14 trees/m^2). However, there was no difference in species richness of the understory between the buffered (approx. 1.3 species/m^2) and unbuffered edge (approx. 2.4 species/m^2). The 30 m wide buffer had been planted 14 years earlier and consisted of 80 different plant species planted 1.8 m apart. The surrounding area consisted of pastures and maintained lawns. The vegetation at each forest edge was sampled using ten 40 m transects, perpendicular to the buffer and the unbuffered forest edge respectively. Each transect contained 10 quadrats (1 × 1 m).

(1) Sonter L. J., Metcalfe D. J. & Mayfield M. M. (2011) Assessing rainforest restoration: the value of buffer strips for the recovery of rainforest remnants in Australia's Wet Tropics. *Pacific Conservation Biology,* 16, 274–288.

8.2. Remove nitogen and phosphorus using harvested products

- We found no evidence of the effects of removing nitrogen and phosphorus using harvested products on forests.

Background
Some ecosystems in human-dominated landscapes have a surplus of nitrogen and phosphorus, mainly resulting from agriculture, industry and traffic. These surpluses can be removed by harvesting forest biomass. However, long term intensive harvesting may reduce soil fertility and hence vegetation productivity.

9. Climate change and severe weather

Key messages

Prevent damage from strong winds

We found no evidence for the effects of preventing damage from strong winds on forests.

9.1. Prevent damage from strong winds

- We found no evidence of the effects of preventing damage from strong winds on forests.

Background

Damage to trees by strong winds may increase tree mortality. However, gaps created by windthrow may also create a more heterogeneous environment and allow light to reach the understory.

10. Habitat protection

Worldwide human activities affect forests in three major aspects: reducing the total area of forest remaining; dividing remaining forest cover into fragments rather than continuous blocks; and changing the structure and composition of the remaining forest. These result in loss of biodiversity. To prevent these threats, forests can be protected.

Key messages

Legal protection of forests

Two site comparison studies in Nigeria and Iran found that legal protection of forest increased tree species richness and diversity or the density of young trees. One replicated, paired site study in Mexico found no effect of forest protection on seed density and diversity of trees and shrubs.

Adopt Protected Species legislation (impact on forest management)

We found no evidence for the effects of adopting Protected Species legislation on forests.

Adopt community-based management to protect forests

Two studies, including one replicated, before-and-after, site comparison, in Ethiopia and Nepal found that forest cover increased more in community-managed forests than in forests not managed by local communities. However, one replicated, site comparison study in Colombia found that deforestation rates in community-managed forests did not differ from deforestation rates in unmanaged forests.

10.1. Legal protection of forests

- Two site comparison studies in Nigeria [3] and Iran [1] found that legal protection of forest increased tree species richness and diversity[3] and the density of young trees[1]. One replicated, paired site study in Mexico[2] found no effect of forest protection on seed density and diversity of trees and shrubs.

Background

Legal protection of forests is considered the best way to protect habitats as it can prevent habitat destruction and biodiversity loss.

A site comparison study in 2005 in temperate broadleaf forest in Iran (1) found that **forest protection increased the density of young trees**. Thirty years after an area was protected, the average number of new trees was higher in protected (530/ha) than in unprotected areas (390/ha). New tree density was monitored in 77 plots (0.1 ha) in one protected and one unprotected forest sites (485 ha each).

A replicated, paired sites study in 1993-2005 in tropical dry forest in Mexico (2) found **no effect of forest protection on seed density or diversity of trees and shrubs**. The total number of tree and shrub seeds was similar in protected (422/m^2) and in disturbed sites (377/m^2), as was the number of species/plot

(18 in both) and species diversity (Shannon's index disturbed: 1.51; protected: 1.66). Two 10 x 20 m treatment plots were replicated at eight sites: disturbed (intensive cattle grazing, fire wood extraction of >160 ton/year; 0.8 ha) and protected (exclusion of human disturbances since 1993). Viable seeds of trees and shrubs were identified using five seed traps in each treatment plot, which were emptied at monthly intervals in 2004-2005.

A site comparison study in tropical moist forest in Nigeria (3) found that **legal protection of forest increased trees species richness and diversity**. The number of tree species observed was 46 vs 24, the number of tree families observed was 21 vs 14, and trees diversity (Shannon's index) was 3.16 vs 3.04 in a protected forest compared with a logged forest. Trees were sampled in eight 20×20 m plots in one protected forest (constituted as strict nature reserve by the forestry research institute of Nigeria) and one logged forest (where active logging activities are in progress).

(1) Alijanpour, A., and Mahmoudzadeh, A. (2007) Investigation and comparison of natural regeneration structure of forest stands in protected and non-protected areas in Arasbaran. *Pakistan Journal of Biological Sciences,* 10, 1697-1702.
(2) Ceccon, E., and Hernández, P. (2009) Seed rain dynamics following disturbance exclusion in a secondary tropical dry forest in Morelos, Mexico. *Revista de biologia tropical*, 57, 257-269.
(3) Adekunle, V.A.J., Olagoke, A.O., and Ogundare, L.F. (2010) Logging impacts in tropical lowland humid forest on tree species diversity and environmental conservation. *Journal of Sustainable Forestry,* 29, 517-538.

10.2. Adopt Protected Species legislation (impact on forest management)

- We found no evidence of the effects of adopting Protected Species legislation on forests.

Background
Protecting individual species may maintain natural ecosystems such as forests. However, the effect is likely to depend on the species in question.

10.3. Adopt community-based management to protect forests

- Two studies from Ethiopia[2] and Nepal[3] (including one replicated, before-and-after, site comparison) found that forest cover increased more in community-managed forests than in forests not managed by local communities. One replicated, site comparison study in Colombia[1] found that deforestation rates in community-managed forests did not differ from deforestation rates in forests that were not managed by local communities, or in uninhabited national parks.

Background
Community-managed forests are forests managed by local communities. They aim to provide a livelihood for local communities while, at the same time, conserving biodiversity. However, a clearly established definition of community-managed forests is missing, and the difference with state and privately managed forests is not always clear (Casse & Millhøj 2013).

Casse T. & Milhøj A. (2013) While waiting for the answer: A critical review of meta-studies of tropical forest management. *Journal of Environmental Management*, 131, 334–342.

A replicated, site comparison study in 1985–2002 in 19 sites in tropical rainforest in Colombia (1) found that **deforestation rates in indigenous reserves did not differ from those in surrounding forests, or those in uninhabited protected national parks**. Deforestation rates in 14 indigenous reserves (0–1.99 %/year) did not differ from the forests surrounding them (0.01–2.89 %/year), or those in five protected national parks (0.02–0.17 %/year). However**,** the deforestation rate in national parks (0.02–0.17 %/year) was lower than in forests surrounding the parks (0.03–0.97 %/year). Deforestation rates were based on satellite images (Landsat, resolution 30 m) of the region taken in 1985, 1992 and 2002. The surrounding forest was defined as the forest within 10 km of the border of the parks and the reserves.

A replicated, before-and-after, site comparison study in 2006–2010 in the Oromia region in Ethiopia (2) found that **adopting community-based forest management increased forest cover**. After two years, the forest cover in community-managed areas increased by 1.5%, compared to a 3.3% decrease in areas that were not managed by local communities. However, in the first year, forest in areas under community-management had a greater deforestation rate (12%) than in areas without community-based management (1.7%), but this was offset by a strong increase in forest cover in the second year (16.9%). The analysis took into account the likelihood that a forest was assigned to community management. From 2007–2009, ninety two areas were brought under community management. Community-based forests were clearly delineated, were monitored by the local community and individual use of forest areas was limited. Forest cover data were based on satellite images (Landsat, resolution 30m) from 2006–2010.

A site comparison study in 1990–2010 in three sites in temperate forest in Dolakha, Nepal (3) found that the **increase in forest cover was higher in community-managed areas than in nearby areas not managed by local communities.** Over a 20-year period, 95% of the non-forested area was converted to forest in community-managed areas. In nearby forests that were not managed by local communities 71% of non-forest area was converted to forest over the same time period. Furthermore, the change from sparse forest (canopy cover 10–40%) to dense forest (canopy cover > 40%) was significantly higher in community-managed forests (62%) than in forests not managed by local communities (60%). At each of the three sites, the community-managed areas and non-community-managed areas were compared. Community-managed forests were managed and monitored by the local communities, according to a management plan they had designed. Tree planting was part of the management plans. Changes in forest cover were monitored using satellite images (Landsat, resolution 30 m) taken in 1990 and 2010.

(1) Armenteras D., Rodriguez N. & Retana J. (2009) Are conservation strategies effective in avoiding the deforestation of the Colombian Guyana Shield? *Biological Conservation,* 142, 1411–1419.
(2) Takahashi R. & Todo Y. (2012). Impact of Community-Based Forest Management on Forest Protection: Evidence from an Aid-Funded Project in Ethiopia. *Environmental Management,* 50, 396–404.
(3) Niraula R.R., Gilani H., Pokharel B.K. & Qamer F.M. (2013) Measuring impacts of community forestry program through repeat photography and satellite remote sensing in the Dolakha district of Nepal. *Journal of Environmental Management*, 126, 20–29.

11. Habitat restoration and creation

The extent and quality of forest habitats is decreasing across the globe, which has significant effects on forest biodiversity. Therefore, it is important not only to conserve, but also to restore destroyed forest ecosystems. Restoring forest biodiversity and the associated ecosystem functioning is crucial for human wellbeing. Selecting suitable tree species assemblages while considering their genetic diversity and functional diversity are important for forest restoration (Aerts & Honnay 2011).

Aerts, R., and Honnay, O. (2011) Forest restoration, biodiversity and ecosystem functioning. *BMC Ecology,* 11, 29.

Key messages - restoration after wildfire

Thin trees after wildfire

Four of five replicated, controlled studies in Spain, Israel, Cananda and the USA found that thinning trees in burnt forest areas increased plant species richness, cover or survival of saplings. One study found thinning decreased plant biomass. One paired-site study in Canada found that logging after wildfire decreased species richness and diversity of mosses.

Plant trees after wildfire

We found no evidence for the effects of planting trees after wildfire on forests.

Sow tree seeds after wildfire

Three studies, including one replicated, randomized, controlled study, in the USA found that sowing herbaceous plant seeds in burnt forest areas decreased the density of tree seedlings or the number and cover of native species. All three found no effect of seeding on total plant cover or species richness.

Remove burned trees

Two replicated, controlled studies in Israel and Spain found that removing burned trees increased total plant species richness or the cover and species richness of some plant species.

Key messages - restoration after agriculture

Restore wood pasture (e.g. introduce grazing)

One replicated paired study in Sweden found that partial harvesting in abandoned wood pastures increased tree seedling density, survival and growth.

Key messages – manipulate habitat to increase planted tree survival during restoration

Use selective thinning after restoration planting

One replicated, paired sites study in Canada found that selective thinning after restoration planting conifers increased the abundance of herbaceous species.

Cover the ground with plastic mats after restoration planting
One replicated study in Canada found that covering the ground with plastic mats after restoration planting decreased the cover of herbecous plants and grasses.

Cover the ground using techniques other than plastic mats after restoration planting
One replicated, randomized, controlled study in the USA found that covering the ground with mulch after planting increased total plant cover.

Apply herbicides after restoration planting
One replicated, randomized, controlled study in the USA found that controlling vegetation using herbicides after restoration planting decreased plant species richness and diversity.

Key messages - restore forest community

Plant a mixture of tree species to enhance diversity
One replicated, randomized, controlled study in Brazil found that planting various tree species increased species richness, but had no effect on the density of new trees. One replicated, controlled study in Greece found that planting native tree species increased total plant species richness, diversity and cover.

Sow tree seeds
One replicated, randomized, controlled, before-and-after study in Brazil found that sowing tree seeds increased the density and species richness of new trees.

Build bird-perches to enhance natural seed dispersal
One replicated, randomized, controlled study in Brazil found that building bird perches increased species richness and abundance of new tree seedlings.

Restore woodland herbaceous plants using transplants and nursery plugs
We found no evidence for the effect of using transplants and nursery plugs on forests.

Use rotational grazing to restore oak savannas
We found no evidence for the effect of using rotational grazing to restore oak savannas.

Water plants to preserve dry tropical forest species
One replicated, controlled study in Hawaii found that watering plants increased the abundance and biomass of forest plants.

Key messages – prevent/encourage leaf litter accumulation

Remove or disturb leaf litter to enhance germination

One of two replicated, controlled studies in Poland and Costa Rica found that removing leaf litter increased understory plant species richness. The two studies found that removal decreased understory plant cover or the density of new tree seedlings.

Encourage leaf litter development in new planting
We found no evidence for the effect of encouraging leaf litter development in new planting on forests.

Key messages - increase soil fertility

Use fertilizer
Six of eight studies, including five replicated, randomized, controlled, in Europe, Brazil, Australia and the USA found that applying fertilizer increased total plant cover, understory plant biomass, size of young trees, biomass of grasses or cover of artificially seeded plant species. Five of the studies found no effect on plant biomass, cover, seedling abundance, tree growth or tree seedling diversity.

Add lime to the soil to increase fertility
One replicated, randomized controlled study in the USA found that adding lime increased vegetation cover.

Add organic matter
One of two studies, including one replicated, randomized, controlled study, in Brazil and Costa Rica found that adding leaf litter increased species richness of young trees. One found it decreased young tree density in artificial forest gaps and both found no effect on the density of tree regenerations under intact forest canopy. One of two replicated, controlled study in Portugal and the USA found that adding plant material increased total plant cover. One found mixed effects on cover depending on plant group.

Use soil scarification or ploughing to enhance germination
Two studies, including one replicated, randomized, controlled study, in Portugal and the USA found that ploughing increased the cover or diversity of understory plants. Two of five studies, including two replicated, randomized, controlled, in Canada, Brazil, Ethiopia and Sweden found that ploughing increased the density of young trees. One found a decrease in density and two found mixed effects depending on tree species. One replicated, before-and-after trial in Finland found that ploughing decreased the cover of plants living on wood surface. One replicated, controlled study in the USA found that ploughing did not decrease the spreading distance and density of invasive grass seedlings.

Use soil disturbance to enhance germination (excluding scarification or ploughing)
Two replicated, controlled studies in Canada and Finland found that disturbance of the forest floor decreased understory vegetation cover.

Use vegetation removal together with mechanical disturbance to the soil

Three studies, including one replicated, randomized, controlled study, in Portugal and France found that vegetation removal together with mechanical disturbance of the soil increased the cover or diversity of understory plants, or density of young trees. One of the studies found it decreased understory shrub cover.

Enhance soil compaction

Two of three studies, including two replicated, randomized, controlled studies in Canada and the USA found that soil compaction increased understory plant cover and density. Two found it decreased tree regeneration height or density and understory plant species richness.

Restoration after wildfire

11.1. Thin trees after wildfire

- Five replicated, controlled studies examined the effects of thinning trees in burnt forest areas. Two studies in Spain[2,5] found that thinning increased **plant species richness**. One in Canada[4] found that it **increased the cover of aspen saplings**. One study in the USA[6] found thinning decreased **plant biomass** and one in Israel[1] found it **decreased mortality of pine seedlings**.
- One paired-site study in Canada[3] found that logging after wildfire decreased species **richness and diversity of mosses**.

Background

After wildfires, thinning is often used as a conservation management practice to reduce fuels (wood) and to reduce future fire risk. This can enhance forest growth, increase its species richness, as well as species and structural diversity.

A replicated, controlled study in 1989-1992 in Aleppo pine (*Pinus halepensis*) forest in Israel (1) found that **thinning decreased the mortality of pine seedlings**. Mortality was higher in control (79%) than in pine thinned (52%) and rockrose (C*istus spp*.) thinned plots (49%), and lowest in plots where both pine and rockrose were thinned (0%). Data were collected in 1992 in four treatment plots (14 × 70 m): no thinning, pine thinned (removing all pine seedlings less than 20-25 cm apart, leaving the tallest ones), rockrose thinned (removing all rockrose seedlings less than 20-25 cm apart, leaving the smaller ones) and pine and rockrose thinned (combined pine and rockrose thinning) in each of five blocks. All blocks were totally burnt down in September 1989. Burned trees were cut down and trunks and smaller twigs removed from the plots in September-November 1990). Thinning was carried out in February 1991.

A replicated, controlled study in 1994-2001 in a Mediterranean Aleppo pine *Pinus halepensis* forest in south east Spain (2) found that **thinning of five year old seedlings increased the number of plant species in one of two study sites but did not affect the total cover of shrubs**. Two years after thinning, in one of the study sites the number of species in thinned (27) was higher than in control (21) plots, while in the other site numbers of species were similar (22 in both). Shrub cover was not affected by thinning at either site (control vs thinning: 85 vs 99%, 60 vs 60%, at each site respectively). Data were collected in June 2001 in three replicates of thinning (leaving a final density of 1,600 trees/ha) and control plots (10 × 15 m). Plots were established in August1999, at each of two sites that were burned by wildfire in August 1994.

A replicated, paired-sites study in 2002-2004 in boreal mixed-wood forest in Alberta, Canada (3) found that **logging after wildfire decreased species richness and diversity of bryophytes**. On burned wood substrate in the first and second years after fire, numbers of species were higher in unlogged (2.6 and 4.6 respectively) than in logged areas (1.6 and 2.6 respectively). Species diversity was higher in unlogged (Shannon's index of diversity: 0.79 and 1.26 respectively) than in logged areas (0.51 and 0.88 respectively). On scorched soil substrate in the first year the number of species and diversity were higher in unlogged (4.6 and 1.36 respectively) than logged areas (3.4 and 1.08 respectively). In the second year results were similar for both number of species: 4.9; diversity: 1.47) and unlogged areas (species: 5.0; diversity: 1.46). Logged and unlogged treatments were applied in each of 24 landscape units of 625 ha in an area burned by wildfire in 2002. Bryophytes were sampled in 72 plots within each treatment in 2003 and 2004.

A replicated, controlled study in 2002-2004 in boreal forest in Alberta, Canada (4) found that **thinning in burned forest increased the cover of trembling aspen *Populus tremuloides* saplings but did not affect plant species richness**. Cover of trembling aspen saplings was higher in thinned (9-11%) than in control plots (4%) while total plant species richness was similar between treatments (16-18 species/plot). Data were collected in 2004 in 40 unthinned control and 74 thinned plots (8 × 8 m), all burned by wildfire in May 2002. Treatments were applied in autumn 2002.

A replicated, controlled study in 1999-2005 in temperate coniferous forest in Spain (5) found that **after wildfire some but not all pruning and thinning treatments increased shrub species richness, but treatments had no effect on shrub species cover**. At one site, the number of shrub species/was lower in untreated (4/10 m transect) than in two out of nine treatments (7) and similar to the other seven treatments (4-7). At the second site numbers of shrub species was lower in untreated (4) than in one out of seven treatments (10) and similar to the other seven treatments (6-8). Shrub cover was similar between treatments at both the first (untreated: 40%; treatments: 30-70%) and second site (untreated: 8%; treatments: 6-30%). In 1999, three untreated and 27 treatment plots (10 × 15 m) were established at one site, and three untreated and 21 treatment plots of similar size were established at a second site. All plots were burned by wildfire in summer 1994. Treatments included different combinations of pruning and thinning (reducing density to 800-1,600 trees/ha) in 1999 and 2004. Data were collected in June 2005 along a 10 m transect in each plot.

A replicated, controlled study in 2004-2006 in temperate mixed forest in Oregon, USA (6) found that **thinning decreased the biomass of live and dead plants in burnt forest areas**. Total dead organic matter was higher in unlogged than in moderate and high-intensity logged plots in both moist (unlogged: 709; moderate-intensity: 355; high-intensity: 244 kg x 10^3/ha) and dry forest units (unlogged: 435; moderate-intensity: 182; high-intensity: 161 kg x 10^3/ha). Total live biomass was higher in unlogged and moderate-intensity than in high intensity treatments in moist forest units (unlogged: 5.6; moderate-intensity: 7.3; high-intensity: 1.6 kg x 10^3/ha). Total live biomass was similar in all treatments in dry forest units (unlogged: 4.9; moderate-intensity: 4.9; high-intensity: 2.8 kg x 10^3/ha). The whole study area was burnt by wildfire in 2002. A 1 ha plot was established in each of eight unlogged, seven moderate-intensity logged (25-75% basal area cut) and six high-intensity logged (>75% basal area cut) moist forest treatment units, as well as three unlogged, three moderate-intensity and three high-intensity logged dry forest treatment units (average 8 ha). Logging occurred in 2004-2006. Data were collected 3-9 months after treatments.

(1) Ne'eman, G., Lahav, H., and Izhaki, I. (1995) Recovery of vegetation in a natural east Mediterranean pine forest on Mount Carmel, Israel as affected by management strategies. *Forest Ecology and Management,* 75, 17-26.
(2) De Las Heras, J., González-Uchoa, A., López-Serrano, F., and Simarro, M. (2004) Effects of silviculture treatments on vegetation after fire in *Pinus halepensis* Mill. woodlands (SE Spain). *Annals of Forest Science,* 61, 661-667.
(3) Bradbury, S.M. (2006) Response of the post-fire bryophyte community to salvage logging in boreal mixedwood forests of northeastern Alberta, Canada. *Forest Ecology and Management,* 234, 313-322.
(4) Macdonald, S.E. (2007) Effects of partial post-fire salvage harvesting on vegetation communities in the boreal mixedwood forest region of northeastern Alberta, Canada. *Forest Ecology and Management,* 239, 21-31.
(5) Moya, D., De Las Heras, J., López-Serrano, F.R., Condes, S., and Alberdi, I. (2009) Structural patterns and biodiversity in burned and managed Aleppo pine stands. *Plant Ecology,* 200, 217-228.
(6) Donato, D.C., Fontaine, J.B., Kauffman, J.B., Robinson, W.D., and Law, B.E. (2013) Fuel mass and forest structure following stand-replacement fire and post-fire logging in a mixed-evergreen forest. *International Journal of Wildland Fire,* 22, 652-666.

11.2. Plant trees after wildfire

- We found no evidence for the effects of planting trees after wildfire on forests.

11.3. Sow tree seeds after wildfire

- Three studies (including one replicated, randomized, controlled study) in the USA[1,2,3] examined the effect of sowing herbaceous plant seeds in burnt forest areas. One found it decreased **the number and cover of native species**[1] and one found it decreased the **density of tree seedlings**[3]. All three found no effect of seeding on total **plant cover**[1,2] or **species richness**[3].

Background
One of the ways to restore trees community after wild is by direct seeding of new trees. This action can also affect the abundance of other plant species and consequently the composition of the whole forest.

A controlled study in 1994-1996 in temperate coniferous forest in Washington State, USA (1) found that **spreading seeds in burnt forest areas decreased the number and cover of native species**. The number of native plants species (unseeded: 17; seeded: 15/m^2) and their cover (unseeded: 41%; seeded: 21%)

were lower in seeded plots. Total plant cover was similar between treatments (unseeded: 41%; seeded: 48%). Thirty-two plots (15 × 15 m) were established in each control (unseeded) and seeded area (seeded in September 1994 with seed mix containing 80% annual grass, 15%, short-lived perennial species and 5% nitrogen-fixing legumes). Both areas (7 ha) burned in July 1994. Data were collected two years after seeding in eight quadrats (1 m^2) in each plot.

A replicated, randomized, controlled study in 2003-2005 in temperate coniferous forest in Washington State, USA (2) found **no effect of spreading seeds in burnt forest areas on total plant cover**. Total plant cover was approximately 55% under both treatments. Seeded species cover was higher in seeded (8%) than in unseeded plots (1.5%). In 2002-2003, seeding (a mixture of perennial graminoids and forbs) and control treatments were randomly assigned to 8-16 plots (6×8 m) established at each of four sites in an area that was burnt by wildfire in summer 2002. Plant cover was measured in summer 2005.

A replicated, controlled study in 2004-2006 in temperate mixed forest in Nevada, USA (3) found that **spreading seeds of a sterile wheat-rye hybrid (triticale) in burnt forest area decreased the density of tree seedlings, but did not affect total species richness or cover of perennial plants**. Numbers of tree seedlings was lower in seeded (14/m^2) than unseeded plots (65/m^2). Total number of species (seeded: 17; unseeded: 18/100 m^2 plot) and total cover of perennial plants (seeded: 24%; unseeded: 28%) were similar between treatments. Data were collected in 2006 in six pairs of seeded (seeded with triticale at ~92 seeds/m^2 in November 2004) and control (unseeded) plots (100 m^2). Sites were in an area that was burnt by wildfire in July 2004.

(1) Schoennagel, T.L., Waller, D.M. (1999) Understory responses to fire and artificial seeding in an eastern cascades abies grandis forest, USA. *Canadian Journal of Forest Research*, 29, 1393-1401.

(2) Dodson, E.K. and Peterson, D.W. (2009) Seeding and fertilization effects on plant cover and community recovery following wildfire in the Eastern Cascade Mountains, USA. *Forest Ecology and Management*, 258, 1586-1593.

(3) Waitman, B.A., Draper, T.M., and Esque, T.C. (2009) The effects of seeding sterile triticale on a native plant community after wildfire in a pinyon pinemountain mahogany woodland. *International Journal of Wildland Fire*, 18, 659-664.

11.4. Remove burned trees

- One replicated, controlled study in Israel[1] found that removing burned trees increased **total plant species richness**. One replicated, controlled study in Spain[2] found that removal increased the **cover and species richness of some plant species**.

Background

In many cases after wildfire, burned trees are removed. The main reasons are that they may provide good wood fuel that increases the intensity of future wildfires. Removing the burned trees is often done by heavy machinery which compresses the soil and may affect the germination and regrowth of plants. Removing the dead organic matter may affect soil mineral content and plant composition. Removing burned trees may also influence the spatial pattern of germination and seedling establishment and change the forest structure and composition.

A replicated, controlled study in 1989-1993 in Aleppo pine *Pinus halepensis* forest in Israel (1) found that **clearing burned trees increased plant species richness**. The number of species was higher in cleared than untreated plots (cleared: 196; twigs remaining: 192; untreated: 185/0.49 ha plot). Data were collected in 1993 in five plots (0.49 ha) of each of three treatments: cleared (burned trees cut down, trunks and smaller twigs removed), twigs (smaller twigs left) and control (untreated). Plots were all in an area totally burnt down in September 1989. Treatments were carried out in September-November 1990.

A replicated, controlled study in 1991-1994 in maritime pine *Pinus pinaster* woodland in Spain (2) found that **removing burned trees increased the cover and species richness of legumes but not of all herbaceous plants, or of the dominant shrub** gum rockrose ***Cistus ladanifer***. Legume cover (removed: 7-29%; control: 3-26%) and species richness (removed: 3-6; control: 2-5/plot) were higher in removal plots. There were no differences between treatments for: total herbaceous cover (removed: 8-47%; control: 3-49%), species richness (removed: 5-16; control: 6-14), gum rockrose cover (removed: 8%-25%; control: 10%-46%) and gum rockrose density (removed: 1-5; control: 5-11/m^2). Data were collected in 12 removal plots (burned trees removed after fire) and 12 control plots (trees not removed, 5 × 5 m). Treatments were three years after the entire study site was burned by wildfire fire in 1991.

(1) Ne'eman, G., Lahav, H., and Izhaki, I. (1995) Recovery of vegetation in a natural east Mediterranean pine forest on Mount Carmel, Israel as affected by management strategies. *Forest Ecology and Management,* 75, 17-26.
(2) Pérez, B. and Moreno, J.M. (1998) Fire-type and forestry management effects on the early postfire vegetation dynamics of a *Pinus pinaster* woodland. *Plant Ecology,* 134, 27-41.

Restoration after agriculture

11.5. Restore wood pasture (e.g. introduce grazing)

- One replicated paired study in Sweden[1] found that partial harvesting in abandoned wood pastures increased tree seedling density, survival and growth.

Background
Wood pastures are semi-open pasture woodlands. Generally, they are maintained by grazing. However, when wood pastures are no longer maintained, other interventions (such as partial harvesting) may be necessary to restore their open character.

A replicated, paired sites, study in 2002–2005 in 25 abandoned wood pastures in southern Sweden (1) found that **abandoned oak *Quercus spp.* wood pastures, subject to partial harvesting had higher oak seedling density, survival and growth than unharvested abandoned wood pastures.** Oak seedling density (harvested: 11,600; unharvested: 3,900 seedlings/ha), survival (harvested: 66%;

unharvested: 44%) and growth (harvested: +2.8 cm; unharvested: -0.8 cm) were higher in harvested compared to unharvested plots. In each of 25 sites (all former wood pastures, abandoned 50–80 years earlier), two 1 ha plots were established. In one of the two plots 26% of the tree basal area was removed during 2002–2003 (all large oaks were retained), whereas the other plot was left unharvested. The number, survival (based on 15 plots) and growth (based on 13 plots) of oak seedlings was recorded using two 100 m transects/plot, containing four subplots (1 × 5 m or 1 × 10 m) each.

(1) Gotmark, F. (2007) Careful partial harvesting in conservation stands and retention of large oaks favour oak regeneration, *Biological Conservation* 140, 349–358.

Manipulate habitat to increase planted tree survival during restoration

11.6. Use selective thinning after restoration planting

- One replicated, paired sites study in Canada[1] found that selective thinning after restoration planting conifers increased the **abundance of herbaceous species and decreased the abundance of trees**.

Background
Harvesting and replanting have substantial effects on forest biodiversity conservation and maintenance of long-term productivity. Selective thinning after restoration planting can increase planted tree establishment success by reducing competition.

A replicated, paired sites study in 1993-1998 in boreal forest in Ontario, Canada (1) found that **cutting of non-coniferous species following planting conifer tree species increased the cover, but not herbaceous species richness; increased species richness but not cover of grasses; decreased the abundance but not species richness of trees**. Percentage cover of herbaceous species was higher in cut than in control plots while their species richness was similar (55 vs 44%, 70 vs 69 species). Species richness of grasses was higher in cut than in control plots while their percentage cover was similar (12 vs 8 species, 15 vs 11%). Species richness and percentage cover of trees 2-10 m were lower in cut than in control plots (15 vs 24 species and 19 vs 29% respectively). For trees 0.5-2 m percentage cover was lower in cut than in control plots while species richness was similar between treatments (50 vs 66%, 39 vs 42 species). Species richness and percentage cover of trees <0.5 m were similar in cut and control plots (44 vs 48 species and 44 vs 43%). Two cutting treatment (chain saw cutting and mechanical brush cutting) and one control plots (4-12 ha) were replicated in four blocks, which had previously been clearcut and planted with white spruce *Picea glauca* and black spruce *Picea mariana* 3-4 years before herbicide treatments. Monitoring was five years after treatment.

(1) Bell, F.W., and Newmaster, S.G. (2002) The effects of silvicultural disturbances on the diversity of seed-producing plants in the boreal mixedwood forest. *Canadian Journal of Forest Research,* 32, 1180-1191.

11.7. Cover the ground with plastic mats after restoration planting

- One replicated study in Canada[1] found that covering the ground with plastic mats after restoration planting decreased the **cover of herbecous plants and grasses**.

Background

Harvesting and replanting have substantial effects on forest biodiversity conservation and maintenance of long-term productivity. Covering the ground using plastic mulch mats can increase the establishment success of planted trees by conserving soil moisture and reducing weed growth and competition.

A replicated study in 1993-1999 in boreal forest in British Colombia, Canada (1) found that plastic **mulch mats decreased the total cover of herbaceous species and grasses in the first three years after treatment, but cover was similar to control plots 5-7 years after treatment**. The total cover of grasses and herbaceous species was lower in plots with mulch mats (39-33%) than in control plots (71-68%) in the first three years, but similar 5-7 years after treatment (mulch: 70-90%; control: 80%). Herbaceous species and grasses were monitored in four 12 × 12 m plots of each treatment: control and covered with 90 × 90 cm plastic mulch mats. The study site had been planted with Douglas-fir *Pseudotsuga menziesii* in 1993.

(1) Harper, G.J., Comeau, P.G., and Biring, B.S. (2005) A comparison of herbicide and mulch mat treatments for reducing grass, herb, and shrub competition in the BC Interior Douglas-fir zone - Ten-year results. *Western Journal of Applied Forestry,* 20, 167-176.

11.8. Cover the ground using techniques other than plastic mats after restoration planting

- One replicated, randomized, controlled study in the USA[1] found that covering the ground with mulch after planting increased **total plant cover**.

Background

Harvesting and replanting have substantial effects on forest biodiversity conservation and maintenance of long-term productivity. Covering the ground using different techniques can increase the establishment success of planted trees by conserving soil moisture and reducing weed growth and competition.

A replicated, randomized, controlled study in 1991-1995 in a degraded temperate coniferous forest in Idaho, USA (1) found that **covering the ground after restoration planting had mixed effects or no effect on vegetation cover depending on material used**. Total plant cover was higher in plots covered with local redtop hay (46-49%) and erosion control blanket (50-54%)

than plots covered with wood-fiber hydro mulch (33-35%) or uncovered (32-35%). Each of four covering treatments: local redtop hay (at 4.5 x 10^3 kg/ha), erosion control blanket (consisting of wood shavings of aspen and alder placed between two plastic nets), wood-fiber hydro mulch (applied at a rate of 1,682 kg/ha) and uncovered was applied in 1991 to four plots (3 × 10 m) at each of two hilltop sites. All plots were planted with western white pine *Pinus monticola* trees, shrubs and grasses before treatments in 1991. Vegetation cover was measured in 1995.

(1) Walsh, J.R., and Redente, E.F. (2011) Comparison of reclamation techniques to re-establish western white pine on smelter-impacted hillsides in Idaho. *Restoration Ecology,* 19, 141-150.

11.9. Apply herbicides after restoration planting

- A replicated, randomized, controlled study in the USA[1] found that controlling vegetation using herbicides after restoration planting decreased **plant species richness and diversity**.

Background
Harvesting and replanting have substantial effects on forest biodiversity conservation and maintenance of long-term productivity. Herbicides have been extensively evaluated for their potential to release planted trees from competing vegetation.

A replicated, randomized, controlled study in 1999-2006 in temperate coniferous forest in Washington State, USA (1) found that **controlling vegetation using herbicides after restoration planting decreased plant species richness and diversity**. Species richness (control: 24; herbicide: 17) and diversity (Simpson's index control: 0.83; herbicide: 0.35) were lower in treated plots. Data were collected in 2006 in two plots (30 × 85 m) of each control and herbicide (annual herbicide applications) treatments in each of four blocks that had been clearcut in 1999. In all plots tree trunks were removed and Douglas-fir *Pseudotsuga menziesii* seedlings were planted in 2000.

(1) Peter, D.H., and Harrington, C. (2009) Six years of plant community development after clearcut harvesting in western Washington. *Canadian Journal of Forest Research,* 39, 308-319.

Restore forest community

11.10. Plant a mixture of tree species to enhance diversity

- One replicated, randomized, controlled study in Brazil[1] found that planting various tree species increased **species richness**, but had no effect on the **density of new trees**.
- One replicated, controlled study in Greece[2] found that planting native tree species increased total plant species richness, diversity and cover.

Background
Direct planting of new trees can be used to restore degraded tree communities. This can also affect the abundance of other plant species and consequently the composition of the whole forest.

A replicated, randomized, controlled before-and-after study in 2004-2005 in subtropical forest in Brazil (1) found that **planting increased species richness, but had no effect on the density of new trees**. The change (after minus before) in number of species was higher in planted plots (planted: 10; unplanted: 0/plot), while the change in stem density was similar between treatments (planted: 1,000; unplanted: 1,000/ha). Data were collected immediately before (January 2004) and one year after treatment (March 2005) in four replicates of adjacent unplanted control and planted (42 seedlings of 18 tree species) plots (10 × 10 m).

A replicated, controlled study in 1998-2003 in a degraded Mediterranean kermes oak *Quercus coccifera* shrubland in Greece (2) found that **planting native pine species increased plant species richness, diversity and cover five years later**. The total number of species (planted: 47; unplanted: 42/plot), number of woody species (planted: 9; unplanted: 7/plot), species diversity (Shannon's index planted: 3.0; unplanted: 2.6) and the total plant cover (planted: 81%; unplanted: 76%) were higher in planted areas. Cover of kermes oak was lower in planted (17%) than in unplanted areas (26%), while the cover of all woody species was similar between treatments (planted: 41%; unplanted: 39%). Planting was in winter 1998. Data were collected five years after planting in one 50 m^2 plot within each 200 m^2 treatment unit. Eighteen units were planted with 30 plants of native Aleppo pine *Pinus halepensis* or stone pine *Pinus Pinea and* 15 were control plots in unplanted areas.

(1) Sampaio, A.B., Holl, K.D., and Scariot, A. (2007) Does restoration enhance regeneration of seasonal deciduous forests in pastures in central Brazil? *Restoration Ecology,* 15, 462-471.
(2) Ganatsas, P., Tsitsoni, T., Tsakaldimi, M., and Zagas, T. (2012) Reforestation of degraded Kermes oak shrublands with planted pines: Effects on vegetation cover, species diversity and community structure. *New Forests*, 43, 1-11.

11.11. Sow tree seeds

- One replicated, randomized, controlled, before-and-after study in Brazil[1] found that sowing tree seeds increased the **density and species richness of new trees**.

Background
Direct seeding of new trees can be used to restore degraded trees community. This can also affect the abundance of other plant species and consequently the composition of the whole forest

A replicated, randomized, controlled, before-and-after study in 2004-2005 in subtropical forest in Brazil (1) found that **sowing tree seeds increased the density and species richness of new trees**. The increase in stem density (seeded: 2,000/ha; unseeded: 1,000) and number of species/plot (seeded: 3; unseeded: 0) was higher in seeded plots. Data were collected immediately before

(January 2004) and one year after treatment (March 2005) in four replicates of adjacent unseeded control and seeded (ten tree species) plots (10 × 10 m).

(1) Sampaio, A.B., Holl, K.D., and Scariot, A. (2007) Does restoration enhance regeneration of seasonal deciduous forests in pastures in central Brazil? *Restoration Ecology,* 15, 462-471.

11.12. Build bird-perches to enhance natural seed dispersal

- One replicated, randomized, controlled study in Brazil[1] found that building perches for birds increased **species richness and abundance of new tree seedlings**.

Background
Building perches for birds can be used to enhance seed dispersal and increase species richness and abundance of tree seedlings. This can help to restore tree communities in degraded forest areas.

A replicated, randomized, controlled study in 2001-2002 in a degraded subtropical *Araucaria* forest in Brazil (1) found that building **bird perches increased species richness and abundance of new seedlings**. Species richness (perches: 0.6-2.0; no perch: 0.2-0.8/m^2) and abundance (perches: 0.7-2.7; no perches: 0.2-1.7) were higher under perches. Data were collected in 2002 in four pairs of perch and control plots (1 × 1 m) in each of 10 blocks randomly located inside a 2 ha area. Perches were 2 m tall with a 16 cm diameter pole and were placed in the centre of each perch plot.

(1) Zanini, L., and Ganade, G. (2005) Restoration of araucaria forest: the role of perches, pioneer vegetation, and soil fertility. *Restoration Ecology,* 13, 507-514.

11.13. Restore woodland herbaceous plants using transplants and nursery plugs

- We found no evidence for the effect of using transplants and nursery plugs on forests.

11.14. Use rotational grazing to restore oak savannas

- We found no evidence for the effect of using rotational grazing to restore oak savannas.

11.15. Water plants to preserve dry tropical forest species

- One replicated, controlled study in Hawaii[1] found that watering plants increased the **abundance and biomass of forest plants**.

Background
Tropical dry forests are among the most endangered and exploited ecosystems in the world. Irrigation may positively affect regeneration in these habitats.

A replicated, controlled study in 1998-2000 in tropical dry forest in Hawaii, USA (1) found that **irrigation increased the abundance and biomass of most forest plants**. Average biomass and density were higher in watered than in control plots for: all species (watered: 355 g/m^2, 28 individuals/m^2; control: 28 g/m^2, 23 individuals/m^2), for native species (watered: 129 g/m^2, 16 individuals/m^2; control: 7 g/m^2, 11 individuals/m^2) and for seeded species (watered: 34 g/m^2, 7 individuals/m^2; control: 1 g/m^2, <1 individuals/m^2). For non-seeded species average biomass was higher in watered (95 g/m^2) than in control plots (6 g/m^2), while density was lower in watered plots (watered: 9; control: 11 individuals/m^2). Thirty two plots (1 m^2) of each treatment: watered (20 litre/plot, three times a week for the first six months, once a week thereafter) and control (not-watered) were established in 1998. Each plot was sown with 60 seeds of shrubs and trees. Plants biomass and density was measured 21 months after treatment.

(1) Cabin, R.J., Weller, S.G., Lorence ,D.H., Cordell, S., and Hadway, L.J. (2002) Effects of microsite, water, weeding, and direct seeding on the regeneration of native and alien species within a Hawaiian dry forest preserve. *Biological Conservation,* 104, 181-191.

Prevent/encourage leaf litter accumulation

11.16. Remove or disturb leaf litter to enhance germination

- One replicated, controlled study in Costa Rica[2] found that leaf litter removal decreased the **density of new tree seedlings**. One replicated, controlled study in Poland[1] found leaf litter removal increased **understory plant species richness** but decreased their **cover**.

Background
A thick litter layer on the forest floor can inhibit seed germination and the development of many forest species. Removing litter can increase germination and consequently biodiversity in forests.

A replicated, controlled study in 1983-1999 in temperate mixed woodland in Poland (1) found that **annual removal of leaf litter increased species richness and cover of mosses after 12 years and temporarily increased vascular plant species richness after 10 years, but decreased vascular plant cover after 13 years**. Species richness and cover of mosses was higher in leaf litter removal plots than in control plots after four years and remained higher until the end of the experiment (average 4-15 years of removal: 8 species, 35% cover;

control: 0 species and 0% cover). Vascular plant cover was lower in leaf litter removal plots than in control plots after 13 years of treatment (average 13-15 years of removal: 55%; control: 85%). Vascular plant species richness was higher in leaf litter removal plots than in control plots after 10 years (average 10-13 years of removal: 17; control: 9 species) and then became similar between treatments after 14 years of treatment (average 14-15 years: 16; control: 11 species). Monitoring was in three pairs of 5 × 5 m plots for two treatments: leaf litter removed (litter raked and removed every year 1983-1998) and controls (litter not removed).

A replicated, controlled study in 1997-1999 in tropical forest in Costa Rica (2) found that **removal of leaf litter decreased the density of new tree seedlings in forest areas, but not in artificial gaps**. The density of new tree seedlings was higher in control (0.5/m^2) than in litter removal plots (0.3/m^2) in forest areas, and similar between treatments in artificial gaps (control: 3.0/m^2; litter removal: 2.7/m^2). In 1997, large gaps (320–540 m^2) were created inside five 40 × 40 m plots (gap plots) by cutting and removing all tree stems ≥5 cm diameter at breast height. Five other similar size plots (non-gap plots) were unmanipulated with respect to canopy cover. Five blocks were established within each plot, each comprised of two 2 × 2 m quadrats one of each of litter removal and a control with no litter removal. Data were taken every two months for one year after treatments.

(1) Dzwonko, Z., and Gawronski, S. (2002) Effect of litter removal on species richness and acidification of a mixed oak-pine woodland. *Biological Conservation,* 106, 389-398.
(2) Dupuy, J.M., and Chazdon, R.L. (2008) Interacting effects of canopy gap, understory vegetation and leaf litter on tree seedling recruitment and composition in tropical secondary forests. *Forest Ecology and Management,* 255, 3716-3725.

11.17. Encourage leaf litter development in new planting

- We found no evidence for the effect of encouraging leaf litter development in new planting on forests.

Increase soil fertility

11.18. Use fertilizer

- Six of eight studies (including five replicated, randomized, controlled) in the USA[1,4,5,7], Finland[6], Brazil[3], Australia[8] and Switzerland[2] found that applying fertilizer increased **total plant cover**[7], **understory plant biomass**[1] , **size of young trees**[4], **relative biomass of grasses** (out of total biomass of all plants)[6] and **cover of plant species that were seeded artificially**[5]. Five of the studies found no effect of applying fertilizer on **plant biomass**[3,6], plant **cover**[5], seedling **abundance**[8], **tree growth**[1,2] and **tree seedling diversity**[3].

Background
Using chemical fertilizers (nitrogen, phosphorus and potassium) increases soil fertility and may therefore enhance tree growth and biodiversity in degraded forest areas. However, it may also enhance growth of other undesired plants.

A controlled study in 1991-1997 in temperate coniferous forest in Louisiana, USA (1) found that **fertilizing increased herbaceous plant biomass but did not affect longleaf pine *Pinus palustris* growth**. Annual herbaceous productivity was higher in fertilized (dry biomass: 472-1795 kg/ha) than in unfertilized plots (452-1088). Average diameter at breast height (30 cm), total height (22 m) and basal area (23-24 m^2/ha) of longleaf pine were similar between treatments. Data were collected in four replicates of 0.64 ha treatment plots: fertilized (50 kg/ha N and 56 kg/ha P applied in April 1991 and May 1997) and unfertilized. Longleaf pine were sampled in February 1996 in four 0.09 ha plots within each treatment. Herbeceous weight was sampled in July 1997 in 12 quadrats (0.02 m^2) within each treatment.

A controlled study in temperate montane forest in 1995-1998 in Switzerland (2) found **no effect of fertilizing on the growth rate of Norway spruce *Picea abies***. Annual increase in height (unfertilized: 11-12 mm; nitrogen addition: 15-16 mm) and diameter (unfertilized: 5 mm; nitrogen addition: 3 mm) was similar between treatments. Monitoring was in 1996-1998 on four trees inside a 1,500 m^2 plot with nitrogen fertilizer added (30 kg N/ha/year starting in 1995) and on five trees in the unfertilized surroundings.

A replicated, randomized, controlled study in 2001-2002 in subtropical *Araucaria* forest in Brazil (3) found **no effect of fertilizing on species richness and abundance of new tree seedlings**. Species richness (fertilized: 0.2-1.9; unfertilized: 0.4-2.0/m^2) and abundance (fertilized: 0.2-2.7; unfertilized: 0.4-2.5/m^2) were similar between treatments. Data were collected in 2002 in two fertilized (nitrogen: 40 kg/ha; phosphorus: 130 kg/ha; potassium: 30 kg/ha) and two unfertilized plots (3 × 3 m) in each of 10 blocks randomly located inside a 2 ha area.

A replicated, randomized, controlled study in 1999-2003 in Piedmont forest in North Carolina, USA (4) found that **applying fertilizer increased the height and diameter of young trees**. At one site trees in fertilized plots were taller (fertilized: 133-137 cm; unfertilized: 103-119 cm) and had greater diameters (fertilized: 13-16 mm; unfertilized: 10-12 mm) than unfertilized plots three years after clearcutting. At the second site there was no difference in tree height (fertilized: 63-71 cm; unfertilized: 63-77 cm) or diameter (fertilized: 9 mm; unfertilized: 9-10 mm) in fertilized and unfertilized plots three years after clearcutting. However, after five years, height (fertilized: 205-212 cm; unfertilized: 154-155 cm) and diameter (fertilized: 21-23 mm; unfertilized: 18-19 mm) were higher in fertilized plots. Data were collected in 2000-2003 in 16 fertilized (phosphorus and potassium at 100 kg/ha each in 1999 and 2001) and 16 unfertilized plots (10 m^2) at each of two sites. The first site was clearcut in 1998-1999, the second in 1996-1997.

A replicated, randomized, controlled study in 2003-2005 in temperate coniferous forest in Washington State USA (5) found that **fertilization increased the cover of plant species that were seeded artificially but did not affect total plant cover**. Seeded species cover was higher in fertilized (12%)

than in unfertilized plots (8%). Total plant cover was approximately 55% under both treatments. In 2002-2003, fertilized (ammonium nitrate and ammonium sulphate) and unfertilized treatments were randomly assigned to 8-16 plots (6 × 8 m) established at each of four sites. Each site had first been covered with a mixture of perennial grass and herbaceous seeds. The area had been burnt by wildfire in summer 2002. Plants cover was measured in summer 2005.

A replicated, randomized, controlled study in 2000-2002 in boreal forest in Finland (6) found that **fertilizing increased the relative weight of grasses (out of total weight of all plant) but not the total weight**. Relative weight of grass was higher in plots treated with 40 and 80 kg/ha of nitrogen (N) than unfertilized plots (unfertilized: 5%; 20 kg N/ha: 7%; 40 kg N/ha: 15%; 80 kg N/ha: 16%). Relative weight of evergreen shrubs (25-40%), deciduous shrubs (53-67%) and herbaceous species (1-7%) and total above ground weight of all plants (12-190 g/m^2) were similar in all treatments. Data were collected in 2002 in eight replicates of four treatments (3 × 3 m): unfertilized, 20, 40 and 80 kg N/ha in a year in 1998-2002, in each of two sites.

A replicated, randomized, controlled study in 2005-2004 in temperate coniferous forest in Washington State, USA (7) found that **fertilizing after wildfire increased plant cover**. Plant cover was higher in plots with low (34-40%) and high (38-45%) input of fertilizer than in unfertilized plots (28-31%). Data were collected in 2005-2006 in 24 plots (4 × 10 m) of each treatment: unfertilized, low fertilizer input and high fertilizer input (0, 56, 112 kg N/ha respectively) in each of eight sites. All sites were burned by wildfire in 2004 and were seeded with different seed mixtures.

A replicated, controlled, randomized study in 1995–2007 in a limestone quarry in Western Australia (8) found that applying **fertiliser over the ground, along with a range of other soil enhancers, did not increase the number of naturally regenerated tree seedlings**. After 12 years, neither fertiliser nor the three soil enhancers increased the number of seedlings in the two experiments (no data provided). Experiment one consisted of four blocks, containing six plots (6 × 10 m). Experiment two consisted of four blocks with four plots (5 × 6 m). Half of the plots in both experiments received fertiliser once (superphosphate: 400 kg/ha and potassium chloride: 100 kg/ha). The plots treated with soil enhancers received all but one of the following treatments: fertiliser tablets, added topsoil, sewage sludge and micronutrients (see paper for details). At the end of the experiments, the number and species of naturally recruited seedlings were recorded for each plot.

(1) Haywood, J.D., Tiarks, A.E., Elliott-Smith, M.L., and Pearson, H.A. (1998) Response of direct seeded Pinus palustris and herbaceous vegetation to fertilization, burning, and pine straw harvesting. *Biomass and Bioenergy*, 14, 157-167.

(2) Schleppi, P., Muller, N., Edwards, P.J., and Bucher, J.B. (2003) Three years of increased nitrogen deposition do not affect the vegetation of a montane forest ecosystem. *Phyton - Annales Rei Botanicae,* 39, 197-204.

(3) Zanini, L., and Ganade, G. (2005) Restoration of araucaria forest: the role of perches, pioneer vegetation, and soil fertility. *Restoration Ecology,* 13, 507-514.

(4) Schuler, J.L., and Robison, D.J. (2006) Stand development and growth responses of 1- and 3-year-old natural upland hardwoods to silvicultural treatments. *Forest Ecology and Management,* 232, 124-134.

(5) Dodson, E.K. and Peterson, D.W. (2009) Seeding and fertilization effects on plant cover and community recovery following wildfire in the Eastern Cascade Mountains, USA. *Forest Ecology and Management,* 258, 1586-1593.
(6) Manninen, O.H., Stark, S., Kytöviita, M.M., Lampinen, L., and Tolvanen, A. (2009) Understory plant and soil responses to disturbance and increased nitrogen in boreal forests. *Journal of Vegetation Science*, 20, 311-322.
(7) Peterson, D.W., Dodson, E.K., and Harrod, R.J. (2009) Fertilization and seeding effects on vegetative cover after wildfire in North-central Washington State. *Forest Science,* 55, 494-502.
(8) Ruthrof K. X., Bell R. & Calver M. (2009) Establishment of Eucalyptus gomphocephala (Tuart) woodland species in an abandoned limestone quarry: effects after 12 years. *Pacific Conservation Biology*, 15, 278–286.

11.19. Add lime to the soil to increase fertility

- One replicated, randomized controlled study in the USA[1] found that adding lime increased **vegetation cover**.

Background
Application of lime (rich in calcium and Magnesium) is used to neutralize soil acidity, and increases activity of soil bacteria. This may increase soil fertility and as a result enhance biodiversity in degraded forest areas.

A replicated, randomized, controlled study in 1991-1995 in a degraded temperate coniferous forest in Idaho, USA (1) found that **adding lime to the soil before restoration planting increased total plant cover**. Total plant cover was higher in lime addition (38-40%) than control plots (17-23%). Control and lime addition treatments (at 11,000 kg/ha) were each applied in 1991 to eight plots (3 × 10 m) at each of two hilltop sites. All plots were fertilized with nitrogen, phosphorus and potassium at 112, 56 and 90 kg/ha respectively. Plots were planted with western white pine *Pinus monticola* trees, shrubs and grasses before treatments in 1991. Data were collected in 1995.

(1) Walsh, J.R., and Redente, E.F. (2011) Comparison of reclamation techniques to re-establish western white pine on smelter-impacted hillsides in Idaho. *Restoration Ecology,* 19, 141-150.

11.20. Add organic matter

- One replicated, randomized, controlled study in Brazil[1] found that leaf litter addition increased **species richness of young trees.** One replicated, controlled study in Costa Rica[2] found leaf litter addition decreased **young tree density** in artificial forest gaps. Both studies found no effect of litter addition on the density of tree regenerations under intact forest canopy.
- One replicated, controlled study in Portugal[4] found that adding plant material to the soil surface increased total plant cover. One replicated, controlled study in the USA[3] found mixed effects on cover depending on understory plant group.

Background
Adding organic matter (plant remains) to the ground increases soil nutrient content and soil moisture. It can also stimulate microbial populations that can

stabilize soil structure. That may increase soil fertility and increase biodiversity in degraded forest areas.

A replicated, randomized, controlled, before-and-after study in 2004-2005 in subtropical forest in Brazil (1) found that **addition of leaf litter increased species richness, but had no effect on the density of new trees**. The change (after minus before) in number of species was higher in litter addition (litter addition: 1; control: 0/plot), while the change in new tree density was similar (litter addition: 1,000; control: 1,000/ha). Data were collected immediately before (January 2004) and one year after treatment (March 2005) in four replicates of adjacent control and leaf litter addition (about 10 cm of dry leaves) plots (10 × 10 m).

A replicated, controlled study in 1997-1999 in tropical forest in Costa Rica (2) found that **addition of leaf litter decreased the density of new tree seedlings in artificial forest gaps, but not under intact forest canopy**. The density of new tree seedlings was higher in control (3.0/m^2) than in litter addition plots (1.7/m^2) inside gaps, but similar between treatments in intact forest (0.5/m^2 in both). In 1997, large gaps (320–540 m^2) were created inside five 40 × 40 m plots (gap plots) by cutting and removing all stems ≥5 cm diameter at breast height. Five other similar size plots (non-gap plots) were unmanipulated with respect to canopy cover. Five blocks were established within each plot, each comprised of two 2×2 m quadrats of each of litter addition and control treatments. Data were taken every two months for one year after treatments.

A replicated, controlled study in 2003-2005 in temperate coniferous forest in Arizona, USA (3) found that **addition of pruned trees had mixed effects on cover of understory plant groups**. Total understory plant cover was higher in the pruned trees treatment in seeded plots in site #1 (pruned trees: 12.4; control: 3.7%). Exotic-plant cover was lower in the pruned trees treatment in seeded (pruned trees: 0.1; control: 1.9%) and non-seeded plots (pruned trees: 0.2; control: 1.6%) in site #2. In both sites, in seeded plots, cover (pruned trees: 1.6-3.9; control: <0.2%) and seed-density (pruned trees: 7-28; control: 2-3/m^2) of grasses was higher in pruned trees treatment plots. Total understory plant cover in site #2 (pruned trees: 11.7-16.3%; control: 11.0-16.1%) and in non-seeded plots in site #1 (pruned trees: 10.7; control: 8.1%) was similar between treatments. Exotic-plant cover in site #1 (pruned trees: 0.0-0.3; control: 0.0%) was similar between treatments. Cover (pruned trees: 0.2-0.9; control: <0.2%) and seed-density (pruned trees: 2-9; control: 0/m^2) of grasses in non-seeded plots were similar between treatments. Two pairs of 1 m^2 treatment plots: control and pruned trees (at 9 kg/m^2) were established within 15 forest openings (0.02-0.05 ha) at each of two sites; one pair of seeded (10 g/m^2 mixture of four native grasses seeded in 2003) and one pair of non-seeded plots. Data were collected in 2005.

A replicated, controlled study in 1998-2007 in Mediterranean oak woodland in Portugal (4) found that **addition of plant material on the soil surface increased total plant cover**. Addition of plant matter (mulching) increased total plant cover to 87% compared with 82% in control plots. In June 1998, mulching and control (no additions) treatments were each applied to three plots (50 × 14 m). In 2007, total plant cover was measured in five 2 × 2 m subplots in each treatment plot.

(1) Sampaio, A.B., Holl, K.D., and Scariot, A. (2007) Does restoration enhance regeneration of seasonal deciduous forests in pastures in central Brazil? *Restoration Ecology,* 15, 462-471.
(2) Dupuy, J.M., and Chazdon, R.L. (2008) Interacting effects of canopy gap, understory vegetation and leaf litter on tree seedling recruitment and composition in tropical secondary forests. *Forest Ecology and Management,* 255, 3716-3725.
(3) Stoddard, M.T., Huffman, D.W., Alcoze, T.M., and Fule, P.Z. (2008) Effects of slash on herbaceous communities in pinyon-juniper woodlands of Northern Arizona. *Rangeland Ecology and Management,* 65, 485-495.
(4) Canteiro, C., Pinto-Cruz, C., Simões, M.P., and Gazarini, L. (2010) Conservation of Mediterranean oak woodlands: Understory dynamics under different shrub management. *Agroforestry Systems,* 82, 161-171.

11.21. Use soil scarification or ploughing to enhance germination

- Two studies (including one replicated, randomized, controlled study) in Portugal[8] and the USA[3] found that ploughing increased the **cover**[8] and **diversity**[3] **of understory plants**.
- Two of three studies (including two replicated, randomized, controlled) in Canada[2,7] and Brazil[5] found that ploughing increased[2,7], and one found it decreased[5] the **density of young trees**. Two replicated, controlled studies in Ethiopia[1] and Sweden[4] found mixed effects of tilling on different tree species.
- One replicated, before-and-after trial in Finland[9] found that ploughing decreased the **cover of plants living on wood surface**.
- One replicated, controlled study in the USA[6] found that ploughing did not decrease the **spreading distance and density of invasive grass seedlings**.

Background

Different soil disturbance treatments are often used to improve degraded soils, mainly before restoration planting. Mechanically scratching or ploughing are the most common techniques. These actions may have mixed effects on different plant groups and may have a significant effect on biodiversity and forest structure.

Other studies on the effects of soil disturbance are discussed in - 'Use vegetation removal together with mechanical disturbance to the soil' and in 'Use soil disturbance to enhance germination (excluding soil scarification or ploughing)'.

A replicated, controlled study in 1992 in Afro-montane forests in Ethiopia (1) found that **ploughing after clearcutting increased seedling establishment of African juniper *Juniperus procera* but not of East African yellowwood *Afrocarpus gracilior* trees**. Seedling density of African juniper (control: 0-13; ploughing: 5-14 individuals/m^2) was higher in ploughing, while density of East African yellowwood (control: 1-5; ploughing: 3-5) was similar between treatments. Data were collected in December 1992 in three pairs of control and ploughing (ploughed to 30 cm depth and raked) subplots (1 × 2 m) in each of nine plots (10 × 10 m) established in a clear-felled site (40 × 40 m) in March-April 1992.

A replicated, randomized, controlled study in 1993-1996 in temperate coniferous forest in Alberta, Canada (2) found that **mechanically scratching the land (scarification) increased the density of white spruce *Picea glauca* seedlings under trembling aspen *Populus tremuloides* canopies**. The density of white spruce under natural regeneration (scarification: 11-16; control: 0 seedlings/ha) and under artificial seeding (scarification: 14-17; control: 0) was higher in scarification plots than controls. Four treatment strips (50 × 6 m): control (undisturbed) and three scarification treatments: light (upper litter layer removed), heavy (humus and litter-layer removed) and heavy with ridge (heavy scarification plus second pass to create a ridge of soil) were established in 1993 in each of three blocks within each of six sites. All sites were dominated by aspen trees. Data were collected in August 1996 in three natural regeneration (not seeded) plots (50 × 100 cm) and three artificially –seeded (100 white spruce seeds in May 1994) plots (50 × 50 cm) in each treatment strip.

A replicated, randomized, controlled study in 1999-2002 in temperate broadleaf forest in Illinois, USA (3) found that **ploughing before reforestation planting increased plant species diversity**. Plant diversity was higher in ploughing (Shannon's index of diversity: 1.7) than in control plots (1.4). Data were collected in 2002 in a 0.5 m^2 plot around each of 60 ash seedlings (planted in 1999) in each control and ploughing (disked to 15 cm depth before planting) treatments (9 × 90 m) replicated in four blocks.

A replicated, controlled study in 1993-2000 in temperate forest in Sweden (4) found that **mechanical soil scarification increased the cover of herbaceous plants; increased the density of young Scots pine *Pinus sylvestris,* downy birch *Betula pubescens* and silver birch *B. pendula* after shelterwood logging; increased pine but decreased birch density after clearcutting; did not affect the density of Norway spruce *Picea abies* seedlings or the cover of grasses and dwarf shrubs**. Density of pine seedlings was higher following scarification in both shelterwood (scarification: 23,000; control: 18,000 seedlings/ha) and clearcut sites (scarification: 6,500; control: 3,000). Birch density was higher in scarification plots in shelterwood sites (scarification: 18,000; control: 3,000) and higher in control plots in clearcut sites (scarification: 7,000; control: 15,000). The density of spruce seedlings was similar between treatments in both shelterwood (17,000-20,000) and clearcut sites (2,500-3,000). Cover of herbaceous plants was higher in scarification plots in both shelterwood (scarification: 9; control: 5%) and clearcut sites (scarification: 11; control: 9%). Cover of grasses and dwarf-shrubs was similar between treatments in both shelterwood (scarification: 19-20; control: 16-18%) and clearcut plots (scarification: 32-35; control 11-12%). Scarification and control treatments were established in each of eight shelterwood (40% of tree volume cut) and eight clearcut plots (0.4 ha). Scarification treatment was applied in 1994-1996, two to14 months after cutting. Data were collected in 2000.

A replicated, randomized, controlled, before-and-after study in 2004-2005 in subtropical forest in Brazil (5) found that **ploughing decreased the density of young trees and had no effect on species richness of new trees**. The change (after minus before) in young tree density was more negative in ploughed plots (ploughing: -4,000; control: 1,000/ha). The number of species/plot (ploughing: -2; control: 0) was similar between treatments. Data were collected immediately before (January 2004) and one year after treatment (March 2005) in four replicates of adjacent control and ploughing (to a 10 cm depth) plots (10 × 10 m).

A replicated, controlled study in 2005–2007 in temperate broadleaf forest in Tennessee, USA (6) found **no effect of soil disturbance on the spreading distance or on the number of invasive grass Japanese stiltgrass *Microstegium vimineum* seedlings**. Average spread distance (disturbed: 13 cm; control: 10 cm) as well number of seedlings (1 to >100 seedlings/m^2) was similar between treatments. Data were collected in 2006-2007 in three disturbed (soil disturbed using a sharpshooter shovel in 2005-2006) and three control plots (1 m^2) in each of three blocks.

A replicated, randomized, controlled study in 2001-2006 in temperate coniferous forest in Alberta, Canada (7) found that **mechanical soil scarification increased the density, but not the height of naturally regenerated pine seedlings**. The density of pine seedlings was higher in scarification plots (scarification: >10,000; control: <1,000 seedlings/ha), while their height was similar between treatments (18-25 cm). Twelve scarification (in winter 2001) and 12 control plots (30 × 30 m) were established in 2002. Density and height of pine seedlings was measured in 2006 in five subplots (10 m^2) within each plot.

A replicated, controlled study in 1998-2007 in Mediterranean oak woodland in Portugal (8) found that **ploughing increased plant cover nine years after treatment**. Total plant cover was higher in ploughing plots (87% in both) than control plots (82%). In June 1998, ploughing (incorporating plant matter into the soil) and control treatments were each applied to three plots (50 × 14 m). Total plant cover was measured in 2007 in five subplots (2 × 2 m) in each treatment plot.

A replicated, before-and-after study in 1998-2000 in boreal Norway spruce *Picea abies* forest in Finland (9) found that **soil scarification after tree felling decreased the cover and number of plant species that living on the surface of wood** (epixylic species). The cover of all epixylic species groups were lower after scarification (vascular plants: before 0.8%, after 0.0%; bryophytes: before 7.0%, after 2.5%; lichens: before 1.1%, after 0.4%). The same was true for the total number of epixylic species (before: 2; after: 1/plot). Epixylic species were monitored before (1999) and after soil scarification in 2000, in approximately 500 plots (200 cm^2) marked on 66 logs in an area that was clear-felled in 1998.

(1) Sharew, H., Legg, C.J., and Grace, J. (1997) Effects of ground preparation and microenvironment on germination and natural regeneration of *Juniperus procera* and *Afrocarpus gracilior* in Ethiopia. *Forest Ecology and Management*, 93, 215-225.

(2) Stewart, J.D., Landhäusser, S.M., Stadt, K.J., and Lieffers, V.J. (2000) Regeneration of White Spruce under Aspen Canopies: Seeding, Planting, and Site Preparation. *Western Journal of Applied Forestry,* 15, 177-182.

(3) Baer, S.G., and Groninger, J.W. (2004) Herbicide and tillage effects on volunteer vegetation composition and diversity during reforestation. *Restoration Ecology,* 12, 258-267.

(4) Karlsson, M., and Nilsson, U. (2005) The effects of scarification and shelterwood treatments on naturally regenerated seedlings in southern Sweden. *Forest Ecology and Management,* 205, 183-197.

(5) Sampaio, A.B., Holl, K.D., and Scariot, A. (2007) Does restoration enhance regeneration of seasonal deciduous forests in pastures in central Brazil? *Restoration Ecology,* 15, 462-471.

(6) Marshall, J.M., and Buckley, D.S. (2008) Influence of litter removal and mineral soil disturbance on the spread of an invasive grass in a Central Hardwood forest. *Biological Invasions,* 10, 531-538.

(7) Landhäusser, S.M. (2009) Impact of slash removal, drag scarification, and mounding on lodgepole pine cone distribution and seedling regeneration after cut-to-length harvesting on high elevation sites. *Forest Ecology and Management*, 258, 43-49.
(8) Canteiro, C., Pinto-Cruz, C., Simões, M.P., and Gazarini, L. (2010) Conservation of Mediterranean oak woodlands: Understory dynamics under different shrub management. *Agroforestry Systems,* 82, 161-171.
(9) Hautala, H., Laaka-Lindberg, S., and Vanha-Majamaa, I. (2011) Effects of retention felling on epixylic species in boreal spruce forests in Southern Finland. *Restoration Ecology*, 19, 418-429.

11.22. Use soil disturbance to enhance germination (excluding soil scarification or ploughing)

- Two replicated, controlled studies from Canada[1] and Finland[2] found that disturbance of the forest floor decreased **understory vegetation cover**.

Background

Soil disturbance treatments are often used to improve degraded soils, mainly before restoration planting. These actions may have mixed effects on different plant groups and may have a significant effect on biodiversity and forest structure.

Studies on the effects of mechanically scratching the soil are discussed in 'Use soil scarification or ploughing to enhance germination'.

A replicated, controlled study in 1999-2000 in boreal forest in Alberta, Canada (1) found that different **forest floor disturbance treatments decreased the cover of herbaceous plants and cranberry *Viburnum edule* and increased the cover of fireweed *Epilobium angustifolium* and the density of *Populus spp.* root-suckers but not its cover**. Cover of fireweed was higher following soil mounding (20%) than in control plots (5%) and intermediate following soil mixing (9%) or removal of the litter layer, 'scalping' (7%). Cover of cranberry was lower in soil mixing (<1%) and soil mounding plots (<1%) than in control plots (2%) and intermediate in litter layer removal plots (1%). Cover of herbaceous plants was lower in soil mixing (1%) and soil mounding plots 2%) than in control (7%) and litter layer removal plots (5%). In litter layer removal plots, *populus spp.* cover (18%) and density of their root-suckers (122,400 stems/ha) were higher than in the other treatments (3-6% cover, 17,500-36,500 stems/ha). In May 1999, four 2x2 m plots of each of four treatments were established within each of six 10 ha forest units. Treatments were: control, soil mixing (mixing the litter layer with the upper 2-3 cm of mineral soil), soil mounding (mineral soil scooped out to form adjacent mound of mineral soil 15 cm high and 1 m in diameter) and litter layer removal ('scalping': litter removal leaving just 2cm of organic matter above the mineral soil). Cover of herbaceous plants was visually estimated in late July 1999. Cover of fireweed, cranberry and *Populus spp*., as well as the root sucker density of *Populus spp*., was evaluated in August 2000.

A replicated, controlled study in 1994-1999 in boreal forest in Finland (2) found that **removal of all vegetation (including bryophytes and lichens) decreased the cover of bryophytes and lichens after five years, while also removing the top soil layer containing organic matter (humus layer) decreased the cover of all understory vegetation**. Total cover of dwarf shrubs, herbaceous plants and grasses was lower with removal of vegetation and the humus layer (<5%) than with removal of just the vegetation (~80%). Cover of bryophytes and lichens was lower with removal of vegetation and the humus layer (25%) than with removal of just the vegetation (50%), and highest in the control (75%). Data were collected in 1999 in 10 plots (0.5 m^2) of each vegetation removal, removal of vegetation and humus layer and control (no removal) plots. Treatments applied in 1994.

(1) Frey, B.R., Lieffers, V.J., Munson, A.D., and Blenis, P.V. (2003) The influence of partial harvesting and forest floor disturbance on nutrient availability and understory vegetation in boreal mixedwoods. *Canadian Journal of Forest Research,* 33, 1180-1188.

(2) Hautala, H., Tolvanen, A., and Nuortila, C. (2003) Recovery of pristine boreal forest floor community after selective removal of understory, ground and humus layers. *Plant Ecology,* 194, 273-282.

11.23. Use vegetation removal together with mechanical disturbance to the soil

- Two studies (including one replicated, randomized, controlled study) in Portugal[2] and France[3] found that vegetation removal together with mechanical disturbance of the soil increased the **cover**[2] and **diversity**[3] **of understory plants.** One of the studies found it also decreased understory **shrub cover**[3].
- One replicated, randomized, controlled study in France[1] found that vegetation removal together with mechanical disturbance of the soil increased the **density of young trees**.

Background

Soil disturbance is often used to improve degraded soils, mainly before restoration planting. In many cases it is applied after clear cutting the existing vegetation. This sequence of actions has mixed effects on different plant groups and may have a significant effect on biodiversity and structure of forests.

Other studies on the effects of soil disturbance are discussed in 'Use soil scarification or ploughing to enhance germination' and in 'Use soil disturbance to enhance germination (excluding scarification or ploughing)'.

A replicated, randomized, controlled study in 2004-2008 in Mediterranean Aleppo pine *Pinus halepensis* woodland in France (1) found that **mechanical cutting of ground vegetation along with mechanical soil disturbance (scarification) increased Aleppo pine seedling density**. Density (seedlings/m^2) in plots with vegetation debris was higher in plots with one (2.8) or double scarification (1.2) than control plots (<0.1). Density in plots with no vegetation debris was highest in double scarification plots (2.8) and higher in one scarification (1.0) than control plots (0.1). Data were collected in January 2008 in 24 plots (14×14 m). There were four replicates of control, one scarification (vegetation cut, litter layer and top soil mechanically scratched in one direction) and double scarification (litter layer and top soil mechanically scratched in two directions) debris plots (vegetation debris scattered in the plot), and 12 plots with the same treatments but with vegetation debris removed. All plots were thinned in 2004 (from 410 to 210 trees/ha). Treatments were applied in 2005.

A replicated, controlled study in 1998-2007 in Mediterranean oak woodland in Portugal (2) found that **cutting shrubs followed by vegetation removal and ploughing increased plant cover nine years after treatment**. Total plant cover was higher in ploughed (87%) than control plots (82%). In June 1998, ploughing and no treatment (control) were each applied to three plots (50 × 14 m). Total plant cover was measured in 2007 in five subplots (2 × 2 m) in each treatment plot.

A replicated, randomized, controlled study in 2005 in Mediterranean Aleppo pine *Pinus halepensis* woodland in France (3) found that **mechanical cutting of ground vegetation together with mechanical soil disturbance (scarification) increased total plant species richness and herbaceous plant cover, decreased shrub cover, but had no effect on total plant diversity**. Herbaceous cover was higher in double scarification than control plots (control: 24; one scarification: 31; double scarification: 34%). Shrub cover was the highest in control and higher in one scarification than in double scarification plots (control: 40; one scarification: 29; double scarification: 20%). Number of species was higher in one and double scarification than control plots (control: 27; one scarification: 35; double scarification: 37species/plot), while diversity was similar between treatments (Shannon's index control: 3.2; one scarification: 3.5; double scarification: 3.6). Data were collected in 2009 in eight replicates of each treatment: control, one scarification (vegetation cut, litter layer and top soil mechanically scratched in one direction) and double scarification (litter layer and top soil mechanically scratched in two directions) plots (14 × 14 m). All plots were thinned in 2004 (from 410 to 210 trees/ha). Treatments were applied in 2005.

(1) Prévosto, B., and Ripert, C. (2008) Regeneration of Pinus halepensis stands after partial cutting in southern France: Impacts of different ground vegetation, soil and logging slash treatments. *Forest Ecology and Management*, 256, 2058-2064.
(2) Canteiro, C., Pinto-Cruz, C., Simões, M.P., and Gazarini, L. (2010) Conservation of Mediterranean oak woodlands: Understory dynamics under different shrub management. *Agroforestry Systems,* 82, 161-171.
(3) Prévosto, B., Bousquet-Mélou, A., Ripert, C., and Fernandez, C. (2011) Effects of different site preparation treatments on species diversity, composition, and plant traits in *Pinus halepensis* woodlands. *Plant Ecology*, 212, 627-638.

11.24. Enhance soil compaction

- Three studies (including two replicated, randomized, controlled) in Canada[1,2] and the USA[3] found that soil compaction decreased **tree regeneration height**[1,2] and **density**[3]. Two of the studies found it increased **understory plant cover**[2] and **density**[3], while one found it decreased **understory plant species richness**[2].

Background
Soil compaction affects soil microclimatic conditions and nutrient availability. This can affect species composition by giving an advantage to early successional understory species.

A replicated, randomized, controlled study in 1995-2000 in boreal forest in British Columbia, Canada (1) found that **soil compaction treatments**

decreased the height of trembling aspen *Populus tremuloides* saplings but not their density. Height of dominant aspen saplings was lower in medium and heavy compaction plots (175 and 170 cm respectively) than in control plots (230 cm). Sapling density was similar between treatments (38,000-39,000 stems/ha). The height of at least 12 dominant aspen saplings and total sapling density were monitored in nine control (no deliberate compaction), nine medium compaction (2 cm impression in soil) and nine heavy compaction (5 cm impression in soil) treatment plots (40×70 m). Treatments were applied in 1995, data were collected in 2000.

A replicated, randomized, controlled study in 1998-2002 in boreal forest in British Columbia, Canada (2) found that **soil compaction increased understory plant cover in debris-removed plots but decreased plant species richness and the height of trembling aspen *Populus tremuloides* saplings in debris remaining plots**. Total cover of shrubs, herbaceous species and mosses was higher in compaction plots with woody debris removal (compaction: 115%; control: 81%). With debris remaining species richness was lower in compaction (17 species/subplot) than control plots (21), as was the maximum height of aspen (compaction: 225 cm; control: 345 cm). There was no difference between compaction treatments and controls for: understory plant cover in debris remaining plots (compaction: 75%; control: 77%); plant species richness in debris removal plots (compaction: 23; control: 22); understory or the maximum height of aspen in debris removal plots (compaction: 110 cm; control: 120 cm). Six compaction (soil depressed by 4–5 cm; 40×70 m) and six control treatment plots were established in 1998-1999. Three of each treatment were assigned as woody debris removal (whole tree harvested, forest floor stripped to expose the soil) and three as debris remaining (trunk only harvested, woody debris left) plots. Under-canopy plants were monitored in 2001 in two subplots (4 m radius). Aspen saplings were measured in 2002 in three subplots within each treatment plot.

A replicated, controlled study in 1994-2003 in temperate broadleaf forest in Missouri, USA (3) found that **soil compaction decreased tree and woody-vine density and increased annual plant density but had no effect on the density of shrubs, perennial herbaceous species and grasses, or on the height of trees or all other plants**. Density of trees was lower in severe compaction than in control plots (control: 5.5; medium compaction: 4.2; severe compaction: 3.2/m^2). Density of woody vines was lower in severe compaction (2.6/m^2) than in medium compaction (4.6) and control plots (4.9). Density of annual herbaceous plants was lower in control (2/m^2) than medium (4.1) and severe compaction plots (3.7). There was no difference between treatments for the density of shrubs (control: 2.5; medium compaction: 3.1; severe compaction: 3.5/m^2), perennial herbaceous species (control: 2.5; medium compaction: 3.1; severe compaction: 2.5/m^2) and grasses (control: 1.2; medium compaction: 1.5; severe compaction: 2.4/m^2), or for the height of trees (control: 2.7; medium compaction: 2.5; severe compaction: 2.3 m) or all other plants (control: 0.6; medium compaction: 0.5; severe compaction: 0.5 m). Data were collected in 2003 in three plots (8 m^2) in each of three replicates of: control (average soil bulk density 1.3 g/cm^3), medium compaction (to 1.7 g/cm^3) and severe compaction (to 1.8 g/cm^3) treatment plots (0.4 ha). Treatments were applied in 1994.

(1) Kabzems, R. and Haeussler, S. (2005) Soil properties, aspen, and white spruce responses 5 years after organic matter removal and compaction treatments. *Canadian Journal of Forest Research,* 35, 2045-2055.
(2) Tan, X., Kabzems, R., and Chang, S.X. (2006) Response of forest vegetation and foliar $\delta^{13}C$ and $\delta^{15}N$ to soil compaction and forest floor removal in a boreal aspen forest. *Forest Ecology and Management,* 222, 450-458.
(3) Ponder Jr., F. (2008) Nine-year response of hardwood understory to organic matter removal and soil compaction. *Northern Journal of Applied Forestry,* 25, 25-31.

12. Actions to improve survival and growth rate of planted trees

This section summarizes the effects of interventions carried out to improve the success of restoration planting. The interventions are similar to those described in earlier sections, but those described the response of natural forests. This section focuses on the responses of planted seedlings.

Key messages

Fence to prevent grazing after tree planting

Four of five studies, including two replicated, randomized, controlled studies, in Finland, Australia, Canada and the USA found that using fences to exclude grazing increased the survival, size or cover of planted trees. Two studies found no effect on survival rate and one found mixed effects on planted tree size.

Use prescribed fire after tree planting

Two of four studies, including one replicated, randomized, controlled study, in Finland, France and the USA found that using prescribed fire after planting increased the survival and sprouting rate of planted trees. One study found fire decreased planted tree size and one found no effect on the size and survival rate.

Mechanically remove understory vegetation after tree planting

Four of five studies, including three replicated, randomized, controlled studies in France, Sweden, Panama, Canada and the USA found no effect of controlling understory vegetation on the emergence, survival, growth rate or frost damage of planted seedlings. One found that removing shrubs increased the growth rate and height of planted seedlings, and another that removing competing herbs increased seedling biomass.

Manage woody debris before tree planting

One replicated, randomized, controlled study in Canada found that removing woody debris increased the survival rate of planted trees. One replicated, controlled study in the USA found mixed effects on the size of planted trees.

Add organic matter after tree planting

Two replicated, randomized, controlled studies in the USA found that adding organic matter before restoration planting increased seedling biomass, but decreased seedling emergence or survival.

Add lime to the soil after tree planting

One of two replicated, randomized, controlled studies in the USA found that adding lime before restoration planting decreased the survival of pine seedlings. One found no effect on seedling growth.

Use fertilizer after tree planting

Two of five studies, including two randomized, replicated, controlled studies, in Canada, Australia, France and Portugal found that applying fertilizer after planting increased the size of the planted trees. Three studies found no effect on the size, survival rate or health of planted trees. One randomized, replicated, controlled study in Australia found that soil enhancers including fertilizer had mixed effects on seedling survival and height.

Use mechanical thinning before or after planting

Five of six studies, including two replicated, randomized, controlled studies, in Brazil, Canada, Finland, France and the USA found that thinning trees after planting increased survival or size of planted trees. One study found mixed effects on survival and size and one found it decreased their density. One replicated study in the USA found that seedling survival rate increased with the size of the thinned area.

Use herbicide after tree planting

Two of three studies, including two replicated, randomized, controlled studies, in Sweden and the USA found that using herbicide increased the size of planted trees. One study found no effect. One replicated, randomized, controlled study in Sweden found no effect of using herbicide on frost damage to seedlings.

Prepare the ground before tree planting

Six of seven studies, including five replicated, randomized, controlled studies, in Canada and Sweden found that ground preparation increased the survival or growth rate of planted trees. One study found no effect of creating mounds on frost damage to seedlings.

Use different planting or seeding methods

Four studies, including one replicated, randomized study, in Australia, Brazil, Costa Rica and Mexico found no effect of planting or seeding methods on the size and survival rate of seedlings. One replicated, controlled study in Brazil found that planting early succession pioneer tree species decreased the height of other planted species.

Cover the ground with straw after tree planting

One replicated, randomized, controlled study in the Czech Republic found that covering the ground with straw, but not bark or fleece, increased the growth rate of planted trees and shrubs.

Use weed mats to protect planted trees

One replicated, controlled study in Hong Kong found no effect of using weed mats on seedling height.

Use tree guards or shelters to protect planted trees

One replicated, randomized, controlled study in the USA found that using light but not dark coloured plastic tree shelters increased the survival rate of planted tree seedlings. One replicated, controlled study in Hong Kong found that tree guards increased tree height after 37 but not 44 months.

Use shading for planted trees
One replicated, controlled study in Panama found that shading increased the survival rate of planted native tree seedlings.

Infect tree seedlings with mycorrhizae
We found no evidence for the effect of injecting tree seedlings with mycorrhizae.

Introduce leaf litter to forest stands
We found no evidence for the effect of introducing leaf litter with beneficial soil biota on planted trees.

Transplant trees
We found no evidence for the effect of transplanting trees to forests.

Use pioneer plants or crops as nurse-plants
We found no evidence for the effect of using pioneer plants or crops as nurse-plants on planted trees.

Reduce erosion to increase seedling survival
We found no evidence for the effect of reducing erosion on planted trees.

Apply insecticide to protect seedlings from invertebrates
One randomized, replicated, controlled study in the USA found that applying insecticide increased tree seedling emergence and survival.

Apply fungicide to protect seedlings from fungal diseases
We found no evidence for the effect of applying fungicies on planted trees.

Improve soil quality after tree planting (excluding applying fertilizer)
Two randomized, replicated, controlled studies in Australia found that different soil enhancers had mixed or no effects on tree seedling survival and height, and no effect on diameter or health.

Water seedlings
One replicated, randomized, controlled study in Spain found that watering seedlings increased or had no effect on seedling emergence and survival, depending on habitat and water availability.

Plant a mixture of tree species to enhance the survival and growth of planted trees
We found no evidence for the effect of planting a mixture of tree species to enhance the survival and growth of planted trees.

12.1. Fence to prevent grazing after tree planting

- Four of five studies (including two replicated, randomized, controlled studies) in Australia[2], Canada[1,4] , Finland[3] and the USA[5] found that using fences to exclude grazing increased the **survival**[2], **size**[2,4,5] and **cover**[1] **of planted trees**. Two studies found no effect on tree survival rate[3,4] and one found mixed effects on planted tree size depending on the structure of the fence.

Background

Grazing by large herbivore can significantly damage new planted trees. Excluding large herbivores from restored forest areas by creating exclosures using wire fences can help the establishment of the planted trees.

A replicated, randomized, controlled study in 1998-2003 in boreal forest in British Columbia, Canada (1) found that **cattle exclusion in rehabilitated forest areas increased the cover of planted lodgepole pine *Pinus contorta* and one of four non-native forage species alsike clover *Trifolium hybridum*, but not of any native species.** Cover of lodgepole pine (ungrazed: 4.5%; grazed: 2%) and alsike clover (ungrazed: 4%; grazed: 2.5%) was higher in ungrazed plots. In contrast, cover of the other three common non-native forage species (1-19%), of the three common native species (0-5%) and of the invasive weed oxeye daisy *Leucanthemum vulgare* (15-27%) was similar between treatments. Data were collected in 2003 in 0.1 ha ungrazed area (fenced with 1.5 m high wire) and 0.2 ha grazed area (230 cow-calf pairs and 20 bulls in May-June and August-September since 1999). Three forest areas were created in mid 1970s and failed to naturally regenerate, planted with 2,450 lodgepole pine seedlings/ha in May 1999.

A replicated, controlled study in 2001-2002 in eucalypt forest in Australia (2) found that **kangaroo exclusion increased planted seedlings biomass and survival rate**. Seedling biomass (excluded 41; control: 27 g dry mass/plot) and survival (excluded: 13/18 plants; control: 10/18 plants) were higher in exclusion plots. Data were collected in winter 2002 in 16 replicates (each planted with a different species) of four exclusion (2.1 m fence in May-June 2001) and four control plots (1.3 × 1.3 m). Each plot was planted with nine plants in August 2001, at each of two rehabilitated bauxite-mine sites.

A replicated, controlled study in 2001-2008 in boreal forest in Finland (3) found that **exclusion of moose *Alces alces* and hares *Lepus spp.* had mixed effects on the height of different tree species, but no effect on their mortality**. The height of Eurasian aspen *Populus tremula* was higher in moose and hare exclusion plots (60 cm) than in moose exclusion and control plots (40 cm). Eurasian aspen mortality was similar in all treatments (17-33%). Height (50-70 cm) and mortality (45-75%) of silver birch *Betula pendula* were similar in all treatments as were the height (45-60 cm) and mortality (0-15%) of rowan *Sorbus aucuparia*. Ten seedlings of each species were planted in 2002-2003 in each of three treatment plots (10 ×15 m): control, moose exclusion (fence mesh size 15 cm) and moose and hare exclusion (fence mesh size 5 cm), replicated in three sites. Treatments were applied in 2002. Data were collected in 2002-2008.

A replicated, controlled study in 1996-2000 in boreal forest in Saskatchewan, Canada (4) found that **herbivore exclusion increase the growth rate but not the survival rate of planted white spruce *Picea glauca* seedlings**. Height increase was greater in two large mammal and large and medium mammal exclusion treatments (25-26 cm) compared to control plots (20 cm). Seedling survival was similar between treatments (75-78%). In 1996, fifteen plots (4 × 8 m) of large mammal exclusion (prevent browsing by moose *Alces alces*, elk and deer), all mammal exclusion (also prevent browsing by snowshoe hares *Lepus americanus*) and control (no exclusion) treatments were established in each of eight blocks. Data were collected in 2000 in four subplots (2 × 2 m) planted with white spruce in June 1996. All plots were harvested (trees >2 m height removed by a feller-buncher) before treatments.

A replicated, randomized, controlled study in 2004-2009 in temperate broadleaf forest in Pennsylvania, USA (5) found that **deer exclusion increased the size of planted northern red oak *Quercus rubra* seedlings**. Seedling height (fenced: 33 cm; unfenced: 16 cm) and root-collar diameter (fenced: 9.5 mm; unfenced: 6.5) were higher in fenced plots. Data were collected in 2009 in four fenced (2.4 m tall wire to exclude deer in 2002-2004) and four unfenced plots (12.5 × 8.5 m) at each of five sites. All plots were partially thinned (shelterwood harvest) within the past 12 years and were planted with northern red oak seedlings in Apr 2004.

(1) Krzic, M., Newman, R.F., Trethewey, C., Bulmer, C.E., and Chapman, B.K. (2006) Cattle grazing effects on plant species composition and soil compaction on rehabilitated forest landings in central interior British Columbia. *Journal of Soil and Water Conservation*, 61, 137-144.
(2) Parsons, M.H., Koch, J., Lamont, B.B., Vlahos, S., and Fairbanks, M.M. (2006) Planting density effects and selective herbivory by kangaroos on species used in restoring forest communities. *Forest Ecology and Management*, 229, 39-49.
(3) Den Herder, M., Kouki, J., and Ruusila, V. (2009) The effects of timber harvest, forest fire, and herbivores on regeneration of deciduous trees in boreal pine-dominated forests. *Canadian Journal of Forest Research,* 39, 712-722.
(4) Milakovsky, B., Frey, B.R., Ashton, M.S., Larson, B.C., and Schmitz, O.J. (2011) Influences of gap position, vegetation management and herbivore control on survival and growth of white spruce (*Picea glauca* (Moench) Voss) seedlings. *Forest Ecology and Management,* 261, 440-446.
(5) Long, R.P., Brose, P.H., and Horsley, S.B. (2012) Responses of northern red oak seedlings to lime and deer exclosure fencing in Pennsylvania. *Canadian Journal of Forest Research,* 42, 698-709.

12.2. Use prescribed fire after tree planting

- Two of four studies (including one replicated, randomized, controlled study) in Finland[1], France[4] and the USA[3,2] found that using prescribed fire after replanting increased the **survival**[4] and **sprouting rate**[2] of **planted trees**. One study found fire decreased **planted tree size** and one found no effect of prescribed fire on the **size**[1] **and survival rate of planted trees**[2].

Background

Prescribed fires are used in the maintenance or restoration of habitats historically subject to occasional 'wildfires' that have been suppressed through management. Using prescribed fires in such habitats can improve the establishment of the planted tree seedlings.

A replicated, controlled study in 2001-2008 in boreal forest in Finland (1) found **no effect of burning on the height or mortality of tree seedlings.** Tree heights were similar in burned and unburned plots for silver birch *Betula pendula* (50-70 cm), rowan *Sorbus aucuparia* (45-50 cm) and Eurasian aspen *Populus tremula* (35-40 cm), as were their mortality rates (54-55%, 8-12% and 27-30% respectively). Ten seedlings of each species were planted in 2002-2003 in each of three burned and three unburned plots (10 ×15 m). The burn treatment was applied in 2001. Data were collected in 2002-2008.

A replicated, controlled study in 1998-2006 in temperate forest in Louisiana, USA (2) found that **prescribed fire decreased the height and basal area of longleaf pine *Pinus palustris* saplings**. Longleaf pine height (March burn: 7.7 m; May burn: 8.7 m; July burn: 8.6 m; control: 9.1 m) and basal area/tree (March burn: 72; May burn: 94; July burn: 92; control: 116 cm^2) were lowest following a burn in March, intermediate and similar following burns in May and July and highest in control plots. Data were collected in 2006 in three plots (0.07 ha) of each treatment: a burn in March, May or July (prescribed burn in 1999, 2001, 2003 and 2005), and a control (untreated since 1998) treatments. Each of the 12 plots was planted with 196 longleaf pine seedlings in 1993-1994.

A replicated, controlled study in 1999-2005 in Mediterranean shrubland in California, USA (3) found that **prescribed fire increased sprouting in planted valley oak *Quercus lobata* saplings without affecting mortality**. The number of new shoots/sapling was higher in burned plots (summer burn: 4.6; spring burn: 4.6; control: 0.8), while the mortality rate was similar between treatments (summer burn: 3%; spring burn: 4%; control: 0%). Data were collected in autumn 2005 in 8-9 blocks (72 m^2) of each of: a summer burn (in 2003), a spring burn (in 2004) and control treatments, with an average of 10 oak trees/block.

A replicated, randomized, controlled study in 2005-2006 in Mediterranean Aleppo pine *Pinus halepensis* woodland in France (4) found that **prescribed burning increased the survival of planted downy oak *Quercus pubescens* and holly oak *Q. ilex* seedlings**. In plots with woody debris, survival of downy oak (control: <0.1; burned: 0.8 seedlings/sowing point) and holly oak (control: 1.0; burned: 2.2) was higher in burned plots. In contrast, in plots without woody debris, survival was similar between treatments for both downy oak (control: 0.0; burned: 0.2) and holly oak (control: 0.7; burned: 1.2). Grass cover was similar between treatments in plots with woody debris (control: 17%; burned: 10%) and without (control: 22%; burned: 16%). Data were collected in 2006 in 16 plots (14 × 14 m). There were four control and four burned (prescribed fire in 2005) treatment plots with woody debris scattered in the plot, and four of each treatment where the woody debris had been manually removed. All plots were thinned in 2004 (from 410 to 210 trees/ha) and in November 2006 holly oak and downy oak were planted with three acorns spaced 1 m apart at each sowing point.

(1) Den Herder, M., Kouki, J., and Ruusila, V. (2009) The effects of timber harvest, forest fire, and herbivores on regeneration of deciduous trees in boreal pine-dominated forests. *Canadian Journal of Forest Research*, 39, 712-722.

(2) Haywood, J. D. (2009) Eight years of seasonal burning and herbicidal brush control influence sapling longleaf pine growth, understory vegetation, and the outcome of an ensuing wildfire. *Forest Ecology and Management,* 258, 295-305.

(3) Holmes, K.A., Veblen, K.E., Berry, A.M., and Young, T.P. (2011) Effects of Prescribed Fires on Young Valley Oak Trees at a Research Restoration Site in the Central Valley of California. *Restoration Ecology,* 19, 118-125.

(4) Prévosto, B., Monnier, Y., Ripert, C., and Fernandez, C. (2011) Diversification of *Pinus halepensis* forests by sowing *Quercus ilex* and *Quercus pubescens* acorns: testing the effects of different vegetation and soil treatments. *European Journal of Forest Research*, 130, 67-76.

12.3. Mechanically remove understory vegetation after tree planting

- Five studies (including three replicated, randomized, controlled studies) in Canada[5], the USA[1], France[6], Panama[3] and Sweden[2] found no effect of controlling understory vegetation on the emergence[1], **survival**[1,3,4,5], **growth rate**[4] or **frost damage in planted seedlings**. However, one found removing competing herbs increased seedling biomass[1].
- One replicated, controlled study in Canada[4] found that removal of sheep laurel shrubs increased the growth rate and height of planted black spruce seedlings.

Background
Mechanical removal of understory vegetation can reduce the competition for resources and help the establishment of planted trees.

A randomized, replicated, controlled study in 1989–1990 in a former arable field, in New Jersey, USA (1) found that removing competing herbs from plots did not increase the emergence or survival of tree of heaven *Ailanthus altissima* seedlings, but did increase seedling biomass. Seedling emergence did not differ between removal and untreated plots (removal: approx. 10; untreated: approx. 9 seedlings/plot). Similarly, seedling mortality was similar between treatments (removal: approx. 10%; untreated: approx. 7%). However, seedling biomass was greater in plots where competing herbs were removed than in untreated plots (no data provided). In 16 plots (0.8 × 1 m), all herbs were clipped at surface level and root connections were severed by driving a spade 30cm in the ground, around the plot's perimeter. The other 16 plots were not clipped. In all plots, 20 seeds had been planted to ensure regeneration.

A replicated, randomized, controlled study in 1988-1995 in boreal forest in Sweden (2) found **no effect of mowing treatment on frost damage to Norway spruce *Picea abies* planted seedlings**. Percentage of seedlings with frost injuries was similar between treatments (6-11% in site 1, 25-35% in site 2). Five blocks of four mowed (ground vegetation cut to <20 cm height when necessary 1989-1993) and four control plots (4×4 m) were established in 1988 in each of two sites. Data were collected in each plot two growing seasons after planting of spruce seedlings.

A replicated, controlled study in 1996-1997 in degraded tropical forest in Panama (3) found **no effect of mowing of invasive grass wild sugarcane *Saccharum spontaneum* on the survival of planted native tree seedlings in abandoned farmlands**. Seedling survival was similar between treatments (once mown: 62%; three mows: 39%; control: 44%). Data were collected in July 1997 in three subplots (1×8 m): once and three times mown (wild sugarcane was hand-cut once or three times during the experiment) and control (untreated), in each of five plots, replicated in five sites. Each subplot was planted with 10 seeds of each of 20 native tree species in July 1996-March 1997.

A replicated, controlled study in 1999-2006 in boreal forest in Quebec, Canada (4) found that **removal of the shrub sheep laurel *Kalmia angustifolia* increased growth rate and height of planted black spruce *Picea mariana* seedlings**. Seedling annual relative growth (removal: 10.3-11.1%; control: 4.7-4.9%) and height (removal: 147-167 cm; control: 57-76 cm) were higher in removal plots. Data were collected in 2005 and 2006 in six removal (sheep laurel removed in August 1999 using glyphosate herbicide and re-sprouting manually clipped from 2000 to 2006) and six plots with no removal within a 0.2 ha area. Each plot was planted with 20 black spruce seedlings at 1 m spacing in June 2000.

A replicated, controlled study in 1996-2000 in boreal forest in Saskatchewan, Canada (5) found **no effect of ground vegetation control treatments on survival or growth rate of planted white spruce *Picea glauca* seedlings**. Survival (75-78%) and height increase (20-26 cm) were similar between treatments. In 1996, fifteen plots (4 × 8 m) of each cutting (all vegetation cut to ground level), crushing (all vegetation and rootstock ground up) and control treatments were established in each of eight blocks. Data were collected in 2000 in four subplots (2 × 2 m) planted with white spruce in June 1996. All plots were harvested (trees >2 m height removed by a feller-buncher) before treatments.

A replicated, randomized, controlled study in 2005 in Mediterranean Aleppo pine *Pinus halepensis* woodland in France (6) found **no effect of mechanical cutting of ground vegetation and scarification treatments on survival of planted downy oak *Quercus pubescens* and holly oak *Q. ilex* seedlings, or on grass cover**. There was no difference between treatments for survival of downy oak (control: <0.1; chopped: 0.2-0.3; one scarification: 0.2-0.6; double scarification: 0.3-0.5 seedlings/sawing point) and holly oak (control: 0.7-1.1; chopped: 1.1-1.6; one scarification: 0.8-1.4; double scarification: 1.2-1.3), or grass cover (control: 16%-17%; chopped: 24%-27%; one scarification: 17%-27%; double scarification: 17%-24%). Data were collected in 2006 in eight replicates of control, chopped (ground vegetation mechanically chopped), one scarification (vegetation chopped, forest floor and top soil loosened in one direction) and double scarification (forest floor and top soil loosened in two directions) plots (14 × 14 m). Treatments were applied in 2005. All plots were thinned in 2004 (from 410 to 210 trees/ha) and seeded in November 2006 with holly oak and downy oak at sowing points of three acorns spaced 1 m apart.

(1) Facelli, J. M. (1994). Multiple indirect effects of plant litter affect the establishment of woody seedlings in old fields. *Ecology*, 75, 1727–1735.

(2) Langvall, O., Nilsson, U., and Örlander, G. (2001) Frost damage to planted Norway spruce seedlings - Influence of site preparation and seedling type. *Forest Ecology and Management*, 141, 223-235.

(3) Hooper, E., Condit, R., and Legendre, P. (2002) Responses of 20 native tree species to reforestation strategies for abandoned farmland in panama. *Ecological Applications,* 12, 1626-1641.

(4) LeBel, P., Thiffault, N., and Bradley, R.L. (2008) Kalmia removal increases nutrient supply and growth of black spruce seedlings: An effect fertilizer cannot emulate. *Forest Ecology and Management,* 256, 1780-1784.

(5) Milakovsky, B., Frey, B.R., Ashton, M.S., Larson, B.C., and Schmitz, O.J. (2011) Influences of gap position, vegetation management and herbivore control on survival and growth of white spruce (*Picea glauca* (Moench) Voss) seedlings. *Forest Ecology and Management,* 261, 440-446.

(6) Prévosto, B., Monnier, Y., Ripert, C., and Fernandez, C. (2011) Diversification of *Pinus halepensis* forests by sowing *Quercus ilex* and *Quercus pubescens* acorns: testing the effects of different vegetation and soil treatments. *European Journal of Forest Research*, 130, 67-76.

12.4. Manage woody debris before tree planting

- One replicated, randomized, controlled study in Canada[2] found that removal of woody debris increased the **survival rate of planted trees.**
- One replicated, controlled study in the USA[1] found mixed effects of removing, chopping and burning woody debris on the size of planted trees.

Background
In forests at higher elevation, where low soil temperatures are limiting factor, the removal of coarse woody debris before restoration planting can affect the establishment of the planted trees.

A replicated, controlled study in 1988-1994 in temperate coniferous forest in Washington State, USA (1) found that **removing, chopping or burning woody debris had mixed effects on the growth of planted Douglas-fir *Pseudotsuga menziesii* and lodgepole pine *Pinus contorta* seedlings**. The average total height growth of both species was lower in cleared than control plots and highest following a spring burn (piled: 61 cm; autumn burn: 66; chopped: 71; pulled off site: 71; piled and burned: 72; control: 75; spring burn: 90). In 1989, seven treatment plots (0.3-3.2 ha) were established in each of four sites: control (untreated); woody debris pulled off site (using a cable system); chopped (debris mechanically chopped); debris piled and burned; debris piled; spring burn (low intensity spring broadcast-burn); autumn burn (low-to-medium intensity autumn broadcast-burn). All plots were clearcut in 1988 and planted with Douglas-fir or lodgepole pine seedlings in 1990. The height of 100 seedlings in each plot was measured at the end of the first and fifth growing seasons.

A replicated, randomized, controlled study in 2001-2006 in temperate coniferous forest in Alberta, Canada (2) found that **woody debris removal decreased the mortality of planted lodgepole pine *Pinus contorta* seedlings**. Mortality of planted seedlings was lower in removal (3%) than control plots (11%). Twelve removal (woody debris removed in winter 2001) and 12 control (woody debris not removed) plots (30 × 30 m) were planted with lodgepole pine (2,000 seedlings/ha) in 2002. The mortality of 20 planted seedlings/plot was monitored in 2003-2006.

(1) Zabowski, D., Java, B., Scherer, G., Everett, R.L., and Ottmar, R. (2000) Timber harvesting residue treatment: Part I. Responses of conifer seedlings, soils and microclimate. *Forest Ecology and Management,* 126, 25-34.
(2) Landhäusser, S.M. (2009) Impact of slash removal, drag scarification, and mounding on lodgepole pine cone distribution and seedling regeneration after cut-to-length harvesting on high elevation sites. *Forest Ecology and Management*, 258, 43-49.

12.5. Add organic matter after tree planting

- Two replicated, randomized, controlled studies in the USA[1,2] found that adding leaf litter or wood-chips before restoration planting **increased seedling biomass**[1], **but decreased seedling emergence**[1] **and survival**[1,2].

Background

Adding wood residuals to the ground increases soil nutrient content and soil moisture. It can also stimulate increases in microbial populations that can stabilize the soil structure. That may help the establishment of planted trees.

A randomized, replicated, controlled study in 1989–1990 in a former arable field in New Jersey, USA (1) found that **adding leaf litter to plots reduced tree of heaven *Ailanthus altissima* seedling emergence and survival, but increased their biomass if competing herbs were present.** Plots with added leaf litter had lower seedling emergence than those without (litter: approx. 6; no litter: approx. 9 seedlings/plot). Seedling mortality in plots with litter was higher (approx. 31%) than in plots without (approx. 7%). Where competing herbs were present, seedling biomass was higher in plots with litter (no data provided). Where competing herbs were absent, biomass was similar with and without litter (no data provided). Sixteen plots (0.8 × 1 m) received dried, cleaned leaf litter from white oak *Quercus alba* at 150 g/m^2, held in place by chicken wire mesh. The remaining 16 plots received no leaf litter. In all plots, 20 seeds had been planted to ensure regeneration.

A replicated, randomized, controlled study in 1991-1995 in a degraded temperate coniferous forest in Idaho, USA (2) found that **addition of wood-chips before restoration planting decreased the survival rate of planted western white pine *Pinus monticola* seedlings**. Survival rate was lower with wood-chips (10-15%) than in untreated plots (72-75%). Untreated and wood-chip addition (at 90,000 kg/ha) treatments were applied in 1991 to eight plots (3 × 10 m) at each of two hilltop sites. All sites were fertilized with nitrogen, phosphorus and potassium at 112, 56 and 90 kg/ha respectively and were planted with western white pine trees, along with shrubs and grasses, before treatments in 1991. Data were collected in 1995.

(1) Facelli, J. M. (1994). Multiple indirect effects of plant litter affect the establishment of woody seedlings in old fields. Ecology, 75, 1727–1735.

(2) Walsh, J.R., and Redente, E.F. (2011) Comparison of reclamation techniques to re-establish western white pine on smelter-impacted hillsides in Idaho. *Restoration Ecology,* 19, 141-150.

12.6. Add lime to the soil after tree planting

- One of two replicated, randomized, controlled studies in the USA[1,2] found that adding lime before restoration planting decreased the **survival of pine seedlings**[1]. The other study found no effect of adding lime on planted **oak seedling growth**[2].

Background

Application of lime (rich in Calcium and Magnesium) is used to neutralize soil acidity and increase activity of soil bacteria. This may increase soil fertility and as result help the establishment of planted trees.

A replicated, randomized, controlled study in 1991-1995 in a degraded temperate coniferous forest in Idaho, USA (1) found that **lime addition before restoration planting decreased the survival of western white pine *Pinus monticola* planted seedlings.** Survival rate was lower with lime (lime: 63-66%) than without lime (72-75%). The two treatments, a control and lime addition (at 11 x 10^3kg /ha) were applied in 1991 to eight plots (3 x 10 m) at each of two hilltop sites. All sites were fertilized with nitrogen, phosphorus and potassium at 112, 56 and 90 kg/ha respectively and were planted with western white pine trees, along with shrubs and grasses, before treatments in 1991. Data were collected in 1995.

A replicated, randomized, controlled study in 2004-2009 in temperate broadleaf forest in Pennsylvania, USA (2) found **no effect of lime addition on the growth of planted seedlings of northern red oak *Quercus rubra***. Seedling height (16-33 cm) and root-collar diameter (6.5-9.5 mm) were similar between treatments. Data were collected in 2009 in two plots (12.5 × 8.5 m) of each treatment: 0, 4.5, 9.0 and 13.5 x 10^3kg /ha lime application rates (applied in May 2004) at each of five sites. All plots were partially thinned (shelterwood harvest) within the past 12 years and were planted with northern red oak seedlings in April 2004.

(1) Walsh, J.R., and Redente, E.F. (2011) Comparison of reclamation techniques to re-establish western white pine on smelter-impacted hillsides in Idaho. *Restoration Ecology,* 19, 141-150.
(2) Long, R.P., Brose, P.H., and Horsley, S.B. (2012) Responses of northern red oak seedlings to lime and deer exclosure fencing in Pennsylvania. *Canadian Journal of Forest Research,* 42, 698-709.

12.7. Use fertilizer after tree planting

- Two replicated, controlled studies in Canada[2] and Portugal[4] found that applying fertilizer after planting increased the **size of the planted trees**. One randomized, replicated, controlled study in Australia[5] found that soil enhancers including fertilizer had a **mixed effect on seedling survival and height**.
- Three studies (including two randomized, replicated, controlled study) in France[1] and Australia[3,5] found no effect of applying fertilizer on the **size and survival rate[1,3] or health[3,5] of planted trees**.

Background
Fertilizer application can be used to improve the establishment of planted trees after forest restoration.

A controlled study in 2001-2003 in Mediterranean type shrubland in France (1) found **no effect of sewage sludge compost application on survival or size of planted downly oak *Quercus pubescens* seedlings**. Seedling annual survival rate (93-100%), height (30.6-33.0 cm) and basal diameter (5.9-6.1 mm) were similar between treatments. About 50 seedlings were planted in May 2001 in each of three 0.13 ha treatment plots: control without compost, 20 and 40 kg compost/seedling. Data were collected in 2002 and 2003.

A replicated, controlled study in 1999-2006 in boreal forest in Quebec, Canada (2) found that **fertilizing increased the height of planted black spruce *Picea mariana* seedlings**. Seedling height was higher in fertilized (76 cm) than control plots (57 cm), while annual relative growth was similar between treatments (4.7-4.9%). Data were collected in 2005 and 2006 in six fertilized (nitrogen, phosphorus, potassium mineral fertilizer at time of planting) and six control plots established within a 0.2 ha area. Each plot was planted with 10 black spruce seedlings at 1 m spacing in June 2000.

A replicated, controlled, randomized study in 1995–2007 in a limestone quarry in Western Australia (3) found that adding fertilizer to the soil **did not increase the survival, height, diameter or health of tree seedlings**. One experiment found that the fertilizer did not affect survival (no data), height (fertilized: 3.2–4.9 m; unfertilized: 4.4–5.2 m), diameter (fertilized: 0.3–12.9 cm; unfertilized: 4.6–7.8 cm) or health class (fertilized: 4–5; unfertilized: 3–4.4) of tuart *Eucalyptus gomphocephala* and limestone marlock *E. decipiens* seedlings. Another experiment found that the fertilizer did not affect survival (no data), height (fertilized: 1.6–6 m; unfertilized: 1.6–6.8 m), diameter (fertilized: 2.8–6.2 cm; unfertilized: 2–7.9 cm) or health (fertilized: 2.3–5; unfertilized: 3.5–4.5) of tuart, limestone marlock and coojong *Acacia saligna* seedlings. Experiment one consisted of four blocks each containing six plots (6 × 10 m). Experiment two consisted of four blocks each with four plots (5 × 6 m). Half of the plots in each experiment were fertilized once (superphosphate: 400 kg/ha and potassium chloride: 100 kg/ha). Five seedlings of each species were planted/plot. After 12 years, the survival, height, diameter and health class (index based on stress, herbivory and nutrient deficiencies, 1: dead; 5: healthy) of all seedlings was assessed.

A replicated, controlled, before-and-after study in 2002-2007 in maritime pine *Pinus pinaster* forest in Portugal (4) found that **fertilizing increased the size of planted maritime pine trees after cutting and chipping but not after cutting and removal of understory vegetation**. After cutting and chipping, growth of maritime pines was greater in fertilized plots compared to unfertilized plots (fertilized: 48 m^3/ha; unfertilized: 32 m^3/ha). After cutting and removal, growth of maritime pine was similar between treatments (fertilized: 31; unfertilized: 25 m^3/ha). Maritime pine trees were measured in July 2002 (immediately after cutting and removal/chipping) and in 2007 in three replicates of fertilized (20 g nitrogen/tree in September 2002) and unfertilized treatments (~800 m^2) in cut and removed (all plants except maritime pine clearcut and removed) and three in cut and chipped plots (all plants clearcut and chipped). Maritime pine trees were planted in 1996 at 1,333 trees/ha.

A randomized, replicated, controlled study in 2008–2009 in two sites in degraded tuart *Eucalyptus gomphocephala* woodlands in Western Australia (5) found that a range of **soil enhancers including fertilizer did not increase tuart seedling health, but had a mixed effect on seedling survival and height.** At one site, seedling survival and height were greater in plots treated with fertilizer tablets (survival: 96%; height 50 cm), fertilizer + moisture retaining chemicals (survival: 80%; height 39 cm), fertilizer + moisture retaining chemicals + metal ion retaining agent (survival: 82%; height 37 cm) than in untreated plots (survival: 54%; height 28 cm). At a second site, there was no effect of treatments on seedling survival or height (see paper for details). Health class was not affected by any of the treatments, at either site. Each site had three blocks each with six plots (6 × 10 m) containing 20 tuart seedlings. Each plot received one of the following treatments: fertilizer tablets, a clay-based amendment, a biological stimulant for soil microbes, fertilizer + moisture retaining chemicals, fertilizer + moisture retaining chemicals + metal ion retaining agent, or was left untreated (for details see study, fertilizer treatments differed). After one year the survival, growth and health of all seedlings was assessed. Seedling health class was based on general vigour, crown density, colour and amount eaten by herbivores.

(1) Larchevêque, M., Montès, N., Baldy, V., and Ballini, C. (2008) Can compost improve Quercus pubescens Willd establishment in a Mediterranean post-fire shrubland? *Bioresource Technology*, 99, 3754-3764.

(2) LeBel, P., Thiffault, N., and Bradley, R.L. (2008) Kalmia removal increases nutrient supply and growth of black spruce seedlings: An effect fertilizer cannot emulate. *Forest Ecology and Management,* 256, 1780-1784.

(3) Ruthrof K. X., Bell R. & Calver M. (2009) Establishment of Eucalyptus gomphocephala (Tuart) woodland species in an abandoned limestone quarry: effects after 12 years. *Pacific Conservation Biology*, 15, 278–286.

(4) Pires, A.L., Xavier, R. (2010) Influence of vegetation management and fertilization on *Pinus pinaster* growth and on understory biomass and composition. *Forest Systems,* 19, 404-409.

(5) Ruthrof K. X., Douglas T. K., Calver M. C., Craig M. D., Dell B. & Hardy G. E. St. J. (2012) The efficacy of soil ameliorants to improve early establishment in trees and shrubs in degraded Eucalyptus gomphocephala woodlands. *Pacific Conservation Biology*, 18, 310–318.

12.8. Use mechanical thinning before or after planting

- Five of six studies (including two replicated, randomized, controlled studies) in Brazil[7], Canada[1,3], Finland[2], France[6] and the USA[4,5] found that thinning trees after planting increased **survival[1,3,6,7] and size[5,6] of the planted trees**. One study found it decreased their **density**[5]. One study found that the effects of thinning on the **size and survival rate of planted trees** vary between species[2].
- One replicated study in the USA[4] found that the **survival rate** of red oak seedlings increased with the size of the thinned area.

Background
Mechanical thinning, that is removal of some trees to reduce the density, is used in restored forest areas to help the establishment of the remaining planted trees by reducing the competition for resources.

A replicated, controlled study in 1993-1997 in boreal forest in Alberta, Canada (1) found that **canopy cutting decreased the mortality of planted white spruce *Picea glauca* seedlings**. Mortality was higher in uncut than i cut treatments (uncut: 22%: partial-cut: 8%-9%; clearcutting: 13%). Seedling height increase (uncut: 14 cm: partial-cut: 29-31 cm; clearcutting: 24 cm) and root-collar diameter (uncut: 5 mm: partial-cut: 7-8 mm; clearcutting: 8 mm) did not differ between treatments. Data were collected in 1997 in one uncut, two partial-cut (residual basal area of 9-16 m^2/ha trembling aspen *Populus tremuloides* and 4 m^2/ha white spruce) and one clearcutplots (150 × 150 m) in each of two blocks. Treatments and seedlings planting were in 1993-1994.

A replicated, controlled study in 2001-2008 in boreal forest in Finland (2) found that **thinning had mixed effects on the height and mortality of different tree species**. In both burned and unburned sites, the height of silver birch *Betula pendula* was higher in thinned and clearcut (130-270 cm) than in unthinned plots (40-70 cm). Birch mortality was lower in thinned (5-10%) than in clearcut and control (25-55%). The height of rowan *Sorbus aucuparia* was similar in all treatments in both burned and unburned plots (40-70 cm). Rowan mortality was higher in clearcut (50%) than in thinned and unthinned plots (5-15%) in unburned, and similar in all treatments in burned sites (5-15%). The height of Eurasian aspen *Populus tremula* was higher in thinned and clearcut (70-80 cm) than in unthinned (30-35 cm) in both burned and unburned. Aspen mortality was higher in unthinned (30%) than in thinned and clearcut (10-15%) in burned, and similar in all treatments in unburned sites (10-30%). Ten seedlings of each species were planted in 2002-2003 in each of three treatment plots (10 ×15 m): clearcut, thinned (50 m^3/ha green-tree retention) and unthinned, replicated in three burned (in 2002) and in three unburned sites (total of 18 plots). Treatments were applied in 2001-2002. Data were collected in 2002-2008.

A replicated, randomized controlled study in 1993-2007 in boreal forest in Ontario, Canada (3) found that **cutting increased the survival rate and size of planted trees**. Survival rate (5-14 years after planting) of white spruce *Picea glauca* (uncut: 36%; cut: 69-74%) and jack pine *Pinus banksiana* (uncut: 6%; cut: 39-52%) was lower in uncut than in the three cut treatments. Height (cm) of white spruce (uncut: 60; partial cut : 180; partial cut and removal: 230; clearcut: 250) and jack pine (uncut: 70; partial cut: 300; partial cut and removal: 400; clearcut: 450) as well as root-collar diameter (cm) of white spruce (uncut: 1; partial cut: 3; partial cut and removal: 5; clearcut: 6) and jack pine (uncut: 1; partial cut: 4; partial cut and removal: 7; clearcut: 9) increased with increasing cutting intensity. In 1993-1994 four treatments: uncut, 50% partial cut, 50% partial cut with removal of residuals after three years, and clearcut were replicated in six blocks (112 × 56 m). Blocks were planted with white spruce and jack pine in 1994. Data were collected in 1998-2007.

A replicated study in 2001-2007 in temperate broadleaf forest in Indiana, USA (4) found that **large gap size increased the survival rate of northern red oak *Quercus rubra* seedlings** compared with medium size gaps, gap size also increased the height and diameter of seedlings planted without containers but not of container-planted seedlings. Survival was higher in large (52-60%) than in medium gap plots (20-41%), but did not differ to small gap plots (33-65%). For bare-root seedling, height (large gaps: 190-210 cm; medium gaps: 125-150 cm; small gaps: 75-100 cm) and diameter (large gaps: 2.0-2.1 cm; medium gaps: 1.6-1.1 cm; small gaps: 0.9-1.0 cm) increased with gap size. Height (190-330 cm) and diameter (2.5-3.2 cm) of container seedlings was similar in all treatments. Four large, four medium and three small gap plots (0.400, 0.100 and 0.024 ha clearcuts, respectively) were established in 2002 and planted with 60, 40 and 20 northern red oak seedlings (both bare-root and container seedlings) respectively. Data were collected five years after planting.

A replicated, randomized, controlled study in 2007-2010 in temperate coniferous forest in Georgia and North Carolina, USA (5) found that **cutting treatments increased the height of planted longleaf pine *Pinus palustris* seedlings, and decreased their density at one of two sites**. At one site, seedling height was lower in uncut (20 cm) than intensively cut (50 cm) and clearcut plots (58 cm) and similar to the last two in intermediate-cut plots (30 cm). The number of new germinants/ha was lower in clearcut (167) than intermediate cut and uncut plots (8,208-10,458) and similar to the other treatments in intensively cut plots (2,319). At a second site, seedling height was lower in intensively cut and uncut plots (30-42 cm) than in clearcut plots (85 cm) and similar to the other treatments in moderate-cut (46 cm). The number of new germinants/ha (32,083-329,167) was similar in all treatments. Monitoring was in May 2010 in four randomly assigned 1 ha treatment plots: uncut; intermediate cut (residual basal area 9 m^2); intensive cut (residual basal area 6 m^2); clearcut. Treatments were replicated seven times at the first site and three times at the second site. All plots were all planted with longleaf pine seedlings in January 2008. Treatments were applied in 2007. In January-April 2010 prescribed burns were conducted in all plots.

A replicated, controlled study in 2007-2010 in Mediterranean Aleppo pine *Pinus halepensis* woodland in France (6) found that **shelterwood cutting increased the height, diameter and survival of planted holly oak *Quercus ilex* and downy oak *Q. pubescens* seedlings**. Seedling height of holly oak (uncut: 10 cm; intermediate cut: 13 cm: intensively cut: 15 cm) and downy oak (uncut: 9 cm; intermediate cut: 8 cm: intensively cut: 9 cm) and stem diameter (uncut: 2.0 and 1.5 mm; intermediate cut: 2.7 and 2.0 mm: intensively cut: 3.3 and 2.4 mm, respectively) differed between all treatments. Number of holly oak seedlings/point was higher in intermediate cut (2.4) than uncut plots (uncut: 2.1) and similar to both in intensively cut plots (2.2). Numbers of downy oak seedlings/point was higher in intermediate cut (2.1) and intensively cut (2.3) than uncut plots (1.3). Data were collected in 2010 in four replicates of uncut, intermediate cut (33% of basal area removed) and intensively cut (66% of basal area removed) treatment plots (25 × 25 m). Plots were established in October and seeded in November 2007 with downy oak and holly oak sowing points of three acorns spaced 1 m apart.

A site comparison study in 1978-1984 in dry tropical forest in Brazil (7) found that **logging increased the survival of newly planted local tree seedlings**. Tree seedling survival was the higher in heavily (64%) than in intermediately logged plots (50%), and the lowest in unlogged plots (41%). Forty seedlings of each of three tree species: *Amburana cearensis, Cedrela fissilis*, and *Sterculia striata* were planted in each of three forest fragments (115-212 ha): heavily logged (in 1997), intermediately logged (in 1996) and unlogged. Mortality was determined one year after planting. Seeds were collected at the study site between June and July and were grown in a greenhouse until planted back in December 2002.

(1) Man, R., and Lieffers, V.J. (1999) Effects of shelterwood and site preparation on microclimate and establishment of white spruce seedlings in a boreal mixedwood forest. *Forestry Chronicle,* 75, 837-844.

(2) Den Herder, M., Kouki, J., and Ruusila, V. (2009) The effects of timber harvest, forest fire, and herbivores on regeneration of deciduous trees in boreal pine-dominated forests. *Canadian Journal of Forest Research,* 39, 712-722.

(3) Man, R., Rice, J.A., and MacDonald, G.B. (2009) Long-term response of planted conifers, natural regeneration, and vegetation to harvesting, scalping and weeding on a boreal mixedwood site, *Forest Ecology and Management,* 258, 1225-1234.

(4) Morrissey, R.C., Jacobs, D.F., Davis, A.S., and Rathfon, R.A. (2010) Survival and competitiveness of Quercus rubra regeneration associated with planting stocktype and harvest opening intensity. *New Forests,* 40, 273-287.

(5) Knapp, B. O., Wang, G.G., Hu, H., Walker, J.L., and Tennant, C. (2011) Restoring longleaf pine (Pinus palustris Mill.) in loblolly pine (Pinus taeda L.) stands: Effects of restoration treatments on natural loblolly pine regeneration. *Forest Ecology and Management,* 262, 1157-1167.
(6) Prévosto, B., Monnier, Y., Ripert, C., and Fernandez, C. (2011) Can we use shelterwoods in Mediterranean pine forests to promote oak seedling development? *Forest Ecology and Management*, 262, 1426-1433.
(7) De-Souza Gomes Guarino, E., and Scariot, A.O. (2012) Tree seedling survival and growth in logged and undisturbed seasonal deciduous forest fragments in central Brazil. *Journal of Forest Research,* 17, 193-201.

12.9. Use herbicide after tree planting

- Two of three studies (including two replicated, randomized, controlled studies) in Sweden[1] and the USA[3,4] found that using herbicide increased the **size of planted trees**[1,3]. One study[4] found no effect on tree size.
- One replicated, randomized, controlled study in Sweden[2] found no effect of using herbicide on **frost damage caused to planted Norway spruce seedlings**.

Background
Herbicides can be used to eliminate competing understory vegetation and to help the establishment of planted trees.

A replicated, randomized, controlled study in 1992-1995 in boreal forest in Sweden (1) found that **applying herbicide increased the biomass of English oak *Quercus robur* seedlings**. Dry weight (g/seedling) of stems (herbicide: 1.25-1.75; untreated: 0.45-0.50) and leaves (herbicide: 0.95-1.20; untreated: 0.25-0.35) were lower in untreated than herbicide plots. Data were collected in 1995 in herbicide and untreated (control) plots (25 m^2) established in summer 1992 in each of six clearcut and six shelterwood (12.5 m^2/ha basal area retained) blocks (cut in 1990). All plots were planted with oak seedlings in November 1992.

A replicated, randomized, controlled study in 1988-1995 in boreal forest in Sweden (2) found **no effect of herbicide treatment on frost damage to planted Norway spruce *Picea abies* seedlings**. The percentage of seedlings with frost injuries was similar between treatments (site 1: 6-13%; site 2: 30-43%). Five blocks of four herbicide (glyphosate emulsion applied directly to the leaves of the ground vegetation whenever necessary through 1989-1993) and four control plots (4 × 4 m) were established in 1988 in each of two sites. Data were collected in each plot two growing seasons after planting of spruce seedlings.

A replicated, randomized, controlled study in 1999-2002 in temperate broadleaf forest in Illinois, USA (3) found that **herbicide treatments during reforestation planting increased seedlings stem volume**. The stem volume index was higher in herbicide treatments before and after seedling emergence (135 and 115 cm^3 respectively) than in control plots (50 cm^3). Stem volume index was calculated in 2002 for 40 ash seedlings (planted in 1999) in each control, after emergence (glyphosate) and before emergence (sulfometuron methyl) herbicide treatments (18 × 30 m) replicated in four blocks. Treatments were applied in 1999.

A replicated, controlled study in 1998-2006 in temperate forest in Louisiana, USA (4) found **no effect of herbicide treatment on the height and basal area of planted longleaf pine *Pinus palustris* trees**. Total cover of understory vegetation was lower in herbicide plots (herbicide: 21%; control 68%). In comparison, longleaf pine height (herbicide: 9.0 m; control 9.1 m) and basal area/tree (herbicide: 12,000 cm^2; control 11,600 cm^2) were similar between treatments. Data were collected in 2006 in three herbicide (application of triclopyr herbicide without intentionally treating herbaceous plants and vine in 1999, 2001, 2003, and 2005) and three control plots (untreated since 1998) of 0.066 ha. Each plots was planted with 196 longleaf pine seedlings in 1993-1994.

(1) Löf, M., Gemmel, P., Nilsson, U., and Welander, N.T. (1998) The influence of site preparation on growth in *Quercus robur* L. seedlings in a southern Sweden clear-cut and shelterwood. *Forest Ecology and Management*, 109, 241-249.
(2) Langvall, O., Nilsson, U., and Örlander, G. (2001) Frost damage to planted Norway spruce seedlings - Influence of site preparation and seedling type. *Forest Ecology and Management*, 141, 223-235.
(3) Baer, S.G., and Groninger, J.W. (2004) Herbicide and tillage effects on volunteer vegetation composition and diversity during reforestation. *Restoration Ecology,* 12, 258-267.
(4) Haywood, J. D. (2009) Eight years of seasonal burning and herbicidal brush control influence sapling longleaf pine growth, understory vegetation, and the outcome of an ensuing wildfire. *Forest Ecology and Management,* 258, 295-305.

12.10. Prepare the ground before tree planting

- Six of seven studies (including five replicated, randomized, controlled studies) in Canada[3,5] and Sweden[1,2,4,6,7] found that ground preparation treatments increased the **survival**[2,3,5,7] **and growth rate**[1,2,6] of planted trees. One study[4] found no effect of creating mounds on frost damage of planted Norway spruce seedlings.

Background
Different soil preparation treatments are used to improve the soil before restoration planting to increase the establishment of planted tree seedlings.

A replicated, randomized, controlled study in 1991-1996 in boreal forest in Sweden (1) found that **site preparation treatments increased the growth rate of planted Scots pine *Pinus sylvestris* seedlings**. Seedling height was higher in scarification (300 mm) than control plots (250 mm), and highest in steamed plots (steamed: 350 mm; burned: 280). Stem basal area (mm^2) differed among all treatments (control: 29; burned: 43; scarification: 55; steamed: 73). In August 1992, five plots (0.6 × 0.6 m) of each control (untreated), burned (ground vegetation and litter burned using a propane burner), scarification (humus layer removed from the mineral soil) and steamed (amount of steam equivalent to 13 L of water evenly sprayed over each plot for 2 minutes) treatments were replicated in 40 blocks that were clearcut in 1991–1992. In June 1993, one scots pine seedling was planted in each plot. Data were collected in 1996.

A replicated, randomized, controlled study in 1986-1996 in boreal forest in Sweden (2) found that **site preparation treatments increased survival and biomass of planted lodgepole pine *Pinus contorta* and Norway spruce *Picea abies* seedlings**. Survival rates were higher with all four site preparation treatments compared to controls for both pine (inverting: 98%; ploughing: 98%:

mounding: 90%: disc-trenching: 86%; control: 72%) and spruce (inverting: 98%; ploughing: 100%: mounding: 96%: disc-trenching: 95%; control: 70%). biomass (g dry weight/seeding) of pine was higher in inverting and ploughing than the other three treatments (inverting: 392; ploughing: 338: mounding: 137: disc-trenching: 143; control: 36) . Biomass of spruce seedlings was higher in inverting than disc-trenching and control treatments, and higher in ploughing and mounding than control plots (inverting: 74; ploughing: 63: mounding: 63: disc-trenching: 32; control: 9). In 1986, five treatments: control (no soil scarification); disc trenching (with powered discs); mounding (with spades); ploughing (with tilt-plough); inverting (tilt-plough made furrows refilled with inverted soil) were randomly replicated eight times. Twenty spruce and 20 pine seedlings were planted in each treatment replicatein 1987. Survival and biomass data were collected 10 and four years after planting respectively.

A replicated, controlled study in 1994-1997 in boreal forest in Alberta, Canada (3) found that **site preparation treatments decreased the mortality of planted white spruce *Picea glauca* seedlings**. Mortality was higher in control than in soil removal and soil mixed plots (control: 23%: soil removal: 7%; soil mixed: 9%). Seedling height increase (control: 23 cm: soil removal: 24 cm; soil mixed: 27 cm) and root-collar diameter (control: 6 mm: soil removal: 7mm; soil mixed: 7 mm) were not different between treatments. Data were collected in 1997 in four plots comprised of three treatment subplots (100 × 33 m): control, soil removal (top 11-13 cm of soil removed) and soil mixed (top 11-13 cm of soil mixed), in each of two blocks. Treatments and seedling planting were undertaken in 1994.

A replicated, randomized, controlled study in 1988-1995 in boreal forest in Sweden (4) found **no effect of mounding treatment on frost damage to planted Norway spruce *Picea abies* seedlings**. The percentage of seedlings with frost injuries (site 1: 6-11%; site 2: 27-38%) was similar between treatments. Spruce seedlings were planted in five blocks of four mounds (50 × 50 cm, 10–20 cm high soil mounds created in the year of planting) and four control plots (4 × 4 m) that were established in 1988 in each of two sites. Data were collected in each plot two growing seasons after planting.

A replicated, randomized, controlled study in 2001-2006 in temperate coniferous forest in Alberta, Canada (5) found that **ground preparation treatments decreased the mortality of planted Lodgepole pine *Pinus contorta* seedlings**. Mortality of planted seedlings was lower in soil mound (1%) and scarification (2%) than untreated plots (11%). Twelve mound, 12 scarification and 12 control plots (30 × 30 m) were established in winter 2001 and planted with lodgepole pine (2,000 seedlings/ha) in 2002. Mortality of planted pines (2003-2006) was monitored by selecting 20 seedlings in each plot.

A replicated, randomized, controlled study in 2006-2008 in boreal forest in Sweden (6) found that soil **mounding increased the biomass of planted English oak *Quercus robur* seedlings, while all site preparation treatments decreased the biomass of ground vegetation**. Dry biomass of English oak was higher in mounding (4-9 g/seedling) than in all other treatments (2-6 g/seedling). Five treatment plots (20×15 m): untreated control; disc trenching; patch scarification; top soil removal and mounding were established in 2006 in each of four blocks (0.35 ha). Data were collected in 2008.

A replicated, controlled study in 2010-2011 in temperate coniferous forest in Sweden (7) found that **site preparation treatments decreased the mortality of planted Douglas-fir *Pseudotsuga menziesii*, but not of Norway spruce *Picea abies* seedlings**. Mortality of Douglas-fir was higher in control than in all site-preparation treatments (control: 40%; scarified: 10%; mound: 6%; inverted: 11%; mixed: 8%). In contrast, mortality of Norway spruce was similar between all treatments (control: 2%; scarified: 4%; mound: 1%; inverted: 2%; mixed: 1%). Forty Norway spruce and 40 Dougla-fir seedlings were planted in May 2010 in four replicates (blocks) of five treatments: control (no treatment); scarified (scarified mineral soil patch); mound (inverted humus turf deposited on the forest floor capped with mineral soil); inverted (inverted humus turf, placed back in the pit covered with mineral soil); mixed (complete mixing of mineral soil and humus). Data were collected in 2010-2011.

(1) Norberg, G., Jaderlund, A., Zackrisson, O., Nordfjell, T., Wardle, D.A., Nilsson, M.-C, and Dolling, A. (1997) Vegetation control by steam treatment in boreal forests: A comparison with burning and soil scarification. *Canadian Journal of Forest Research,* 27, 2026-2033.
(2) Örlander, G., Hallsby, G., Gemmel, P., and Wilhelmsson, C. (1998) Inverting improves establishment of *Pinus contorta* and *Picea abies* - 10-year results from a site preparation trial in northern Sweden. *Scandinavian Journal of Forest Research*, 13, 160-168.
(3) Man, R., and Lieffers, V.J. (1999) Effects of shelterwood and site preparation on microclimate and establishment of white spruce seedlings in a boreal mixedwood forest. *Forestry Chronicle,* 75, 837-844.
(4) Langvall, O., Nilsson, U., and Örlander, G. (2001) Frost damage to planted Norway spruce seedlings - Influence of site preparation and seedling type. *Forest Ecology and Management*, 141, 223-235.
(5) Landhäusser, S.M. (2009) Impact of slash removal, drag scarification, and mounding on lodgepole pine cone distribution and seedling regeneration after cut-to-length harvesting on high elevation sites. *Forest Ecology and Management*, 258, 43-49.
(6) Löf, M., and Birkedal, M. (2009) Direct seeding of *Quercus robur* L. for reforestation: The influence of mechanical site preparation and sowing date on early growth of seedlings. *Forest Ecology and Management*, 258, 704-711.
(7) Wallertz, K., and Malmqvist, C. (2013) The effect of mechanical site preparation methods on the establishment of Norway spruce (*Picea abies* (L.) Karst.) and Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco) in southern Sweden. *Forestry,* 86, 71-78.

12.11. Use different planting or seeding methods

- Four studies (including one replicated, randomized study) in Australia[2], Brazil[3], Costa Rica[4] and Mexico[1] found no effect of planting or seeding methods on the **size and survival rate of seedlings**.
- One replicated, controlled study in Brazil[5] found that planting early succession pioneer tree species **decreased the height of other planted species.**

Background
Different planting methods have been developed to improve the establishment of the planted trees.

A replicated study in tropical forest in Mexico (1) found that **seeding method had no effect on mahogany *Swietenia macrophylla* seedling survival and height**. Survival rate (32-35%) and height (20-22 cm) did not differ between treatments. Twenty-five plots (0.2 ha) were each seeded with 300 mahogany seeds using two seeding methods: seeds dropped from 1 m height and seeds planted into 3 cm deep holes. Data were collected 12 months after seeding.

A replicated, controlled study in 2001-2002 in eucalypt forest in Australia (2) found **no effect of planting density on planted seedlings biomass and survival rate**. Seedling weight (low density 34.2; high density: 34.5 g dry mass/plot) and survival (leaving plant/18 plants) (low density 11.3; high density: 11.4) were similar between treatments. Data were collected in winter 2002 in 16 replicates (each planted with a different species) of eight plots: four planted at low-density (with nine plants) and four at high-density (0.5 and 0.1 m between plants respectively) in August 2001, at each of two rehabilitated bauxite-mine sites.

A replicated, randomized study in 2004-2005 in tropical forest in Paraná, Brazil (3) found **no difference between manual or mechanical planting on seedlings growth rate**. One year after planting, the height (manual: 88 cm; mechanical: 59 cm) and height relative growth (manual: 0.88 cm/cm; mechanical: 0.98 cm/cm) were similar between treatments. Two treatments: manual planting (holes dug manually; seedlings wrapped in polyethylene bags) and mechanical planting (soil prepared with a rotary tiller attached to a tractor; seedlings in polypropylene tubes) were established in three random 20 × 20 m plots. Seedlings were planted in July 2004 and were measured one month and 13 months after planting.

A replicated, paired-sites study in 2004-2008 in tropical forest in Costa Rica (4) found **no effect of planting method on seedling survival, height, and canopy area**. For the four planted species, there was no difference between patch and plantation treatments for: survival (*Terminalia amazonia:* 70-75%; *Vochysia guatemalensis*: 74-77%; *Erythrina poeppigiana*: 84-87%; *Inga edulis*: 95-97%), height increase (*T. amazonia:* 1.8-2.3 m; *V. guatemalensis*: 2.5-3.0 m; *E. poeppigiana*: 3.7-3.9 m; *I. edulis*: 4.2-4.8 m) and canopy area (*T. amazonia:* 2-4 m^2; *V. guatemalensis*: 5-7 m^2; *E. poeppigiana*: 7-8 m^2; *I. edulis*: 25-30 m^2). Twelve pairs of two treatments (50 × 50 m): patch (two small, two medium, and two large patches each planted with 5, 13 and 25 seedlings of the four species respectively) and plantation (313 seedlings of the four species planted throughout) were established in 2004-2005. Data were collected three years after planting.

A replicated, controlled study in 2004-2008 in tropical forest in Brazil (5) found that **planting early succession pioneer tree species decreased the height of the other planted species**. Seedlings were taller in plots without pioneer species (no pioneers planted: 269 cm; pioneers planted: 243 cm). In 2004, thirty six plots (50 × 50 m) were planted with 4–9 tree-seedlings/m^2. Half of the plots were planted with seedlings of 120 non-pioneer tree species and half of the plots were also planted with 2-4 pioneer species. Data were collected in 2008. All plots were cleared and then abandoned in 1990.

(1) Negreros-Castillo, P., and Hall, R.B. (1996) First-year results of partial overstory removal and direct seeding of mahogany (*Swietenia macrophylla*) in Quintana Roo, Mexico. *Journal of Sustainable Forestry,* 3, 65-76.

(2) Parsons, M.H., Koch, J., Lamont, B.B., Vlahos, S., and Fairbanks, M.M. (2006) Planting density effects and selective herbivory by kangaroos on species used in restoring forest communities. *Forest Ecology and Management*, 229, 39-49.

(3) Bruel, B.O., Marques, M.C.M., and Britez, R.M. (2010) Survival and growth of tree species under two direct seedling planting systems. *Restoration Ecology,* 18, 414-417.
(4) Holl, K.D., Zahawi, R.A., Cole, R.J., Ostertag, R., and Cordell, S. (2011) Planting seedlings in tree islands versus plantations as a large-scale tropical forest restoration strategy. *Restoration Ecology*, 134, 155-165.
(5) Massad, T.J., Chambers, J.Q., Rolim, S.G., Jesus, R.M., and Dyer, L.A. (2011) Restoration of pasture to forest in Brazil's Mata Atlantica: the roles of herbivory, seedling defenses, and plot design in reforestation. *Restoration Ecology,* 19, 257-267.

12.12. Cover the ground with straw after tree planting

- One replicated, randomized, controlled study in the Czech Republic[1] found that covering the ground with straw, but not bark or fleece, increased the **growth rate of planted trees and shrubs**.

Background
Covering the ground with straw during tree planting can decrease evaporation and germination of competing species, and thus increase the establishment rate of planted trees.

A replicated, randomized, controlled study in 2000-2005 in temperate broadleaf woodland in the Czech Republic (1) found that **covering the ground with straw mulch increased the growth rate of planted trees and shrubs, but covering with bark or fleece mulches did not**. Five years after planting, the average growth of trees and shrubs was higher with straw mulch (226 cm) than with bark mulch (124 cm), fleece mulch (100 cm) or no mulch (80 cm). In 2000, seedlings of a mixture of 128 tree and 190 shrub species were planted in each of four 600 m^2 plots. Each plot was divided into four 150 m^2 subplots that were randomly assigned to four treatments: straw mulch (planting and covering the whole area with a 0.3 m layer of straw mulch); bark mulch (planting in a previously established grassland and applying a 0.2 m layer of fresh bark in 0.4 m rows); fleece mulch (planting in a previously established grassland and applying layers of 0.4 m wide synthetic fleece in rows); no mulch (planting in a previously established grassland). Plants were measured in 2005.

(1) Dostálek, J., Weber, M., Matula, S., and Frantík, T. (2007) Forest stand restoration in the agricultural landscape: The effect of different methods of planting establishment. *Ecological Engineering,* 29, 77-86.

12.13. Use weed mats to protect planted trees

- One replicated, controlled study in Hong Kong[1] found no effect of using weed mats on thick-leaved **oak seedling height**.

Background
Weed mats can be used to eliminate competing understory vegetation and help the establishment of planted trees.

A replicated, controlled study in 1999-2002 in a degraded tropical forest in Hong Kong (1) found **no effect of using weed mats on thick-leaved oak**

***Cyclobalanopsis edithae* seedling height**. Seedling height was similar in control and weed mat treatments after 37 months (control: 58; weed mats: 57 cm) and after 44 months (control: 83; weed mats: 85 cm). Fifteen oak seedlings were planted in each of four replicates (rows) of each control (no treatment after planting) and weed mats (0.4 × 0.4 m hessian cloth around each seeding) treatments. Seedlings were planted in June 1999 and observed for approximately 3.5 years.

(1) Lai, P.C.C., and Wong, B.S.F. (2005) Effects of tree guards and weed mats on the establishment of native tree seedlings: implications for forest restoration in Hong Kong, China. *Restoration Ecology,* 13, 138-143.

12.14. Use tree guards or shelters to protect planted trees

- One replicated, randomized, controlled study in the USA[1] found that using light but not dark coloured plastic tree shelters increased the **survival rate of planted tree seedlings**. One replicated, controlled study in Hong Kong[2] found that tree guards increased **tree height** after 37 but not 44 months.

Background
Tree guards can be used to protect planted tree seedlings from browsing, cold etc. and increase their establishment chances.

A replicated, randomized, controlled study in 1996-1997 in temperate coniferous forest in Colorado, USA (1) found that **tree shelters increased the survival rate of Engelmann spruce *Picea engelmannii* seedlings**. The survival rate of Engelmann spruce was higher with the three light-coloured tree shelters (95-99%) than the controls (70%) and the lowest with the dark brown shelters (5%). Four replicates of each of five treatments were randomly assigned to 20 plots in each of three blocks (0.5 ha): four colours of recycled polyethylene plastic tree shelters (31 cm height and 9 cm diameter), ranging from nearly clear to brown, and a control (using materials from within the site, e.g. logs, stumps, shrubs, rocks, to protect seedlings). In August-September 1996, 25 seedlings were planted in each plot (total of 1,500 seedlings). Data were collected in 1997.

A replicated, controlled study in 1999-2002 in a degraded tropical forest in Hong Kong (2) found that using **tree guards increased the height of thick-leaved oak *Cyclobalanopsis edithae* seedlings, also covered with weed mats**, but only in the first three years. After 37 months, seedling height was greater with tree guards (80 cm) than control (57 cm). However, after 44 months, there was no difference between treatments (control: 85; tree guards: 96 cm). Fifteen oak seedlings were planted in each of four replicate (rows) of each tree guards (45 cm high plastic tree guard) and control treatments. All rows were covered with weed mats (0.4 × 0.4 m hessian cloth around each seeding). Seedlings were planted in June 1999 and observed for approximately 44 months.

(1) Jacobs, D.F., and Steinbeck, K. (2001) Tree shelters improve the survival and growth of planted Engelmann spruce seedlings in southwestern Colorado. *Western Journal of Applied Forestry*, 16, 114-120.

(2) Lai, P.C.C., and Wong, B.S.F. (2005) Effects of tree guards and weed mats on the establishment of native tree seedlings: implications for forest restoration in Hong Kong, China. *Restoration Ecology,* 13, 138-143.

12.15. Use shading for planted trees

- One replicated, controlled study in Panama[1] found that shading increased the **survival rate of planted native tree seedlings**.

Background
Shading newly planted trees can decrease evaporation or germination of competing species, and thus increase establishment chances of planted trees.

A replicated, controlled study in 1996-1997 in degraded tropical forest in Panama (1) found that **shading increased the survival of planted native tree seedlings.** The proportion of seedlings that survived out of those that germinated was higher in 75% shaded (74%) and 95% shaded plots (78%) than unshaded plots (39%). Data were collected in July 1997 in three treatment subplots (1×8 m): 95% shaded, 75% shaded and unshaded, in each of five plots, replicated in five sites. Each subplot was planted with 10 seeds of each of 20 native tree species in July 1996-March 1997. In all plots wild sugarcane *Saccharum spontaneum* was hand-cut three times during the experiment.

(1) Hooper, E., Condit, R., and Legendre, P. (2002) Responses of 20 native tree species to reforestation strategies for abandoned farmland in panama. *Ecological Applications,* 12, 1626-1641.

12.16. Infect tree seedlings with mycorrhizae

- We found no evidence for the effect of inoculating tree seedlings with mycorrhizae.

12.17. Introduce leaf litter to forest stands

- We found no evidence for the effect of introducing leaf litter to introduce beneficial soil biota on planted trees.

12.18. Transplant trees

- We found no evidence for the effect of transplanting trees on planted trees.

12.19. Use pioneer plants or crops as nurse-plants

- We found no evidence for the effect of using pioneer plants or crops on planted trees.

12.20. Reduce erosion to increase seedling survival

- We found no evidence for the effect of reducing erosion on seedling survival.

12.21. Apply insecticide to protect seedlings from invertebrates

- One randomized, replicated, controlled study in the USA[1] found that applying insecticide increased tree seedling emergence and survival

Background
Insecticides can be used to eliminate invertebrate herbivores and to help the establishment of planted trees.

A randomized, replicated, controlled study in 1989–1990 in a former arable field, in New Jersey, USA (1) found that using insecticides increased the number of emerging tree of heaven *Ailanthus altissima* seedlings and seedling survival. Plots treated with insecticide had greater seedling emergence (approx. 13 seedlings/plot) than untreated plots (approx. 9 seedlings/plot). Additionally, seedling mortality was lower in plots treated with insecticide (approx. 3 %) than in untreated plots (approx. 7%). Sixteen plots (0.8 × 1 m) were treated with Carbaril dust 5%, dosed at 5 g active ingredient/m^2. The other 16 plots were not treated with insecticide. In all plots, 20 seeds had been planted to ensure sufficient regeneration.

(1) Facelli, J. M. (1994) Multiple indirect effects of plant litter affect the establishment of woody seedlings in old fields. *Ecology*, 75, 1727–1735.

12.22. Apply fungicide to protect seedlings from fungal diseases

- We found no evidence for the effect of applying fungicides to planted trees.

Background
Fungicides can be used to reduce or eliminate fungi or fungal spores and protect planted seedlings from fungal diseases.

12.23. Improve soil quality after tree planting (excluding applying fertilizer)

- One of two randomized, replicated, controlled studies in Australia[1, 2] found that different soil enhancers had mixed effects on tree seedling survival and height, but no effect on tree seedling health[1]. The other found that combinations of soil enhancers did not increase seedling survival, height, diameter or health[2].

Background
Using soil enhancers (excluding fertilizers) can improve soil properties and may therefore enhance tree growth and biodiversity in degraded forest areas. However, it may also enhance growth of other undesired plants.

A randomized, replicated, controlled study in 1995–2007 in a limestone quarry in Western Australia (1) found that **adding a variety of soil enhancers together to the soil did not increase the survival, height, diameter or health of tree seedlings**. Experiment one found that three soil enhancers did not affect survival (no data), height (soil enhancers: 0.06–7m; untreated: 4.4–5.2 m), diameter (soil enhancers: 0.3–12.9 cm; untreated: 4.6–7.8 cm) or health class (soil enhancers: 2–5; untreated: 3–4.4) of tuart *Eucalyptus gomphocephala* and Limestone Marlock *E. decipiens* seedlings. Experiment two found that adding three soil enhancers with fertiliser tablets did not affect survival (no data), height (soil enhancers: 1.6–6 m; untreated: 1.6–6.8 m), diameter (soil enhancers: 1.5–6.5 cm; untreated: 2–7.9 cm) or health (soil enhancers: 2.3–5; untreated: 3.5–4.5) of tuart, Limestone Marlock and coojong *Acacia saligna* seedlings. Experiment one consisted of four blocks each containing six plots (6 × 10 m). Experiment two consisted of four blocks each with four plots (5 × 6 m). In experiment one, treated plots received all but one of the following treatments: fertiliser tablets, added topsoil, sewage sludge and micronutrients (details see paper). In experiment two, treated plots received all four treatments. Half the plots in each experiment received one application of broadcast fertilizer (superphosphate: 400 kg/ha and potassium chloride: 100 kg/ha). Five seedlings of each species were planted/plot. After 12 years, the survival, height, diameter and health class (index based on stress, herbivory and nutrient deficiencies, 1: dead; 5: healthy) of all seedlings was assessed.

A randomized, replicated, controlled study in 2008–2009 in two sites in degraded tuart *Eucalyptus gomphocephala* woodlands in Western Australia (2) found that **adding soil enhancers (other than fertilizers) had mixed effects on tuart seedling survival and height but no effect on seedling health**. At one site, seedlings were taller in plots treated with a biological stimulant for soil microbes (approx. 106 cm) than in untreated plots (approx. 82), but smaller where a clay-based amendment was added (approx. 63 cm; data taken from a graph). No other treatments had an effect on height (see paper for additional data). None of the treatments increased survival rate. At a second site, seedlings height and survival were higher in plots with fertilizer + moisture retaining chemicals (survival: 80%; height 39 cm) or fertilizer + moisture retaining chemicals + metal ion retaining agent (survival: 82%; height 37 cm) than in untreated plots (survival: 54%; height 28 cm). However, they did not differ from plots treated with fertilizer tablets (survival: 96%; height 50 cm). None of the other treatments had an effect on survival or height. None of the treatments had an effect on plant health class at either site. Each site had three blocks each with six plots (6 × 10 m) containing 20 tuart seedlings. Each plot received one of the following treatments: fertilizer tablets, a clay-based amendment, a biological stimulant for soil microbes, fertilizer + moisture retaining chemicals, fertilizer + moisture retaining chemicals + metal ion retaining agent, or was left untreated (for details see study, fertilizer treatments differed). After one year the survival, growth and health of all seedlings was assessed. Seedling health class was based on general vigour, crown density, colour and amount eaten by herbivores.

(1) Ruthrof K. X., Bell R. & Calver M. (2009) Establishment of Eucalyptus gomphocephala (Tuart) woodland species in an abandoned limestone quarry: effects after 12 years. *Pacific Conservation Biology*, 15, 278–286.
(2) Ruthrof K. X., Douglas T. K., Calver M. C., Craig M. D., Dell B. & Hardy G. E. St. J. (2012) The efficacy of soil ameliorants to improve early establishment in trees and shrubs in degraded Eucalyptus gomphocephala woodlands. *Pacific Conservation Biology*, 18, 310–318.

12.24. Water seedlings

- One replicated, randomized, controlled study in Spain[1] found that watering tree seedlings increased survival during a dry summer but only increased the survival of some species during a wet summer, depending on the habitat. Watering increased or had no effect on seedling emergence depending on habitat and water availability.

Background
Watering seedlings can relieve drought stress in young trees and may therefore enhance their survival and growth. However, it may also enhance growth of undesired competing plants.

A replicated, randomized, controlled study in 2004–2004 in one site in southern Spain (1) found **that watering sown seeds increased or had no effect on the emergence and survival of trees, depending on species.** In 2003, watering increased seedling emergence during the first two years for three of six species: holly oak *Quercus ilex* (watered: 43–57%; unwatered: 37–49%), common whitebeam *Sorbus aria* (watered: 7–34%; unwatered: 4-24%) and common yew *Taxus baccata* (watered: 23–39%; unwatered: 1–9%). In 2004, at the same site, watering increased seedling emergence during the first two years for two of five species: Scotch pine in open areas and woodland (watered: 37–42%; unwatered: 22–24%), but not shrubland, and whitebeam in woodland (watered: 32%; unwatered: 12–28%), but not open or shrubland. In 2003, watering increased seedling survival of all six species during the first two years across all three habitats (watered: approx. 0–100%; unwatered: approx. 0–90%; data taken from graphs). In 2004, watering did not increase seedling survival, except for Scotch pine in open areas and scrubland (watered: 2–39%; unwatered: 0–5%) and whitebeam in open areas (watered: 0–40%; unwatered: 0–19%). Experiments contained 180 (in 2003) and 90 (in 2004) plots (20 × 20 cm) across three open areas, scrubland and woodlands. Each plot contained the following seeds: 5 holm oak; 5 Pyrenean oak *Q. pyrenaica*; 15 Italian maple *Acer opalus*; 15 common whitebeam; 15 Scotch pine; 10 common yew (only in 2003). Half of the plots were watered during the year of sowing (2 l of water to a 30 × 30 cm area, frequency unknown). The year 2004 was wetter than 2003 (mean volumic water content of unwatered plots in summer: 2003: 8%; 2004: 18%).

(1) Mendoza I., Zamora R. & Castro J. (2009) A seeding experiment for testing tree-community recruitment under variable environments: Implications for forest regeneration and conservation in Mediterranean habitats. *Biological Conservation*, 142, 1491–1499.

12.25. Plant a mixture of tree species to enhance survival and growth of planted trees

- We found no evidence for the effect of planting a mixture of tree species to enhance the survival and growth of planted trees

Background

Competition within species is generally stronger than competition between species (Connel 1983). Therefore, planting multiple tree species together may reduce the competition between planted seedlings and increase overall seedling survival.

Connel J.H (1983) On the prevalence and relative importance of interspecific competition: evidence from field experiments. The American Naturalist, 122, 661–695.

13. Education and awareness raising

Key messages

Raise awareness amongst the general public through campaigns and public information

We found no evidence for the effect of raising awareness amongst the general public through campaigns and public information on forests.

Provide education programmes about forests

We found no evidence for the effect of providing education programmes about forests on forest habitat.

13.1. Raise awareness amongst the general public through campaigns and public information

- We found no evidence for the effect of raising awareness amongst the general public through campaigns and public information on forests.

13.2. Provide education programmes about forests

- We found no evidence for the effect of providing education programmes about forests on forest habitat.

Appendix 1. Interventions in the Forest Synopsis. Interventions are classified into three groups: (a) interventions included in all searches and relevant articles found and summarized (*n*=60), (b) interventions included in all searches but no relevant articles found (*n*=17), and (c) interventions not included in the search of the two specialist journal or keyword search (*n*=38).

Threat (level 1)	**Threat (level 2)**	**action**
Residential and commercial development	Housing and urban areas	Compensate for woodland removal with compensatory planting
		Incorporate existing trees or woods into the landscape of new developments
		Provide legal protection of forests from development
	Tourism and recreation areas	Create managed paths/signs to contain disturbance
		Re-route paths, control access or close paths
		Use warning signs to prevent fires
		Adopt ecotourism
Agriculture	Livestock farming	Use wire fences within grazing areas to exclude livestock from specific forest sections
		Remove livestock grazing in forests
		Reduce the intensity of livestock grazing in forests
		Shorten livestock grazing period or control grazing season in forests
		Provide financial incentives not to graze
Transport and service corridors		Maintain/create habitat corridors
Biological resource use	Thinning and wood harvesting	Thin trees within forests
		Log/remove trees within forests
		Remove woody debris after timber harvest
		Use shelterwood harvest instead of clearcutting
		Use partial retention harvesting instead of clearcutting
		Use summer instead of winter harvest
		Adopt continuous cover forestry
		Use brash mats during harvesting to avoid soil compaction
	Harvest forest products	Sustainable management of non-timber forest products
		Adopt Certification
	Firewood	Provide fuel efficient stoves
		Provide paraffin stoves
Natural system modification	Changing fire frequency	Use prescribed fire
		Use herbicides to remove understory vegetation to reduce wildfires
		Mechanically remove understory vegetation to reduce wildfires
	Water management	Recharge groundwater to restore wetland forest
		Construct water detention areas to slow water flow and

		restore riparian forests
		Introduce beavers to impede water flow in forest watercourses
	Change disturbance regime	Use clearcutting to increase understory diversity
		Use shelterwood harvesting
		Use group-selection harvesting
		Use herbicides to thin trees
		Thin trees by girdling (cutting rings around tree trunks)
		Use thinning followed by prescribed fire
		Reintroduce large herbivores
		Pollard trees (top cutting or top pruning)
		Coppice trees
		Halo ancient trees
		Adopt conservation grazing of woodland
		Retain fallen trees
		Imitate natural disturbances by pushing over trees
Invasive and other problematic species	Invasive plants	Manually/mechanically remove invasive plants
		Use herbicides to remove invasive plant species
		Use grazing to remove invasive plant species
		Use prescribed fire to remove invasive plant species
	Native plants	Manually/mechanically remove native plants
	Large herbivores	Use wire fences to exclude large native herbivores
		Use electric fencing to exclude large native herbivores
		Control large herbivore populations
		Use fencing to enclose large herbivores (e.g. deer)
	Medium-sized herbivores	Control medium-sized herbivores
	Rodents	Control rodents
	Birds	Control birds
Pollution		Maintain/create buffer zones
		Remove nitrogen and phosphorus using harvested products
Climate change and severe weather		Prevent damage from strong winds
Habitat protection		Legal protection of forests
		Adopt Protected Species legislation (impact on forest management)
		Adopt community-based management to protect forests
Habitat restoration	Restoration after wildfire	Thin trees after wildfire
		Plant trees after wildfire
		Sow tree seeds after wildfire
		Remove burned trees
	Restoration after agriculture	Restore wood pasture (e.g. introduce grazing)
	Manipulate	Use selective thinning after restoration planting

	habitat to increase planted tree survival during restoration	Cover the ground with plastic mats after restoration planting
		Cover the ground using techniques other than plastic mats after restoration planting
		Apply herbicides after restoration planting
		Water plants to preserve dry tropical forest species
	Restore forest community	Plant a mixture of tree species to enhance diversity
		Sow tree seeds
		Build bird-perches to enhance natural seed dispersal
		Use rotational grazing to restore oak savannas
		Restore woodland herbaceous plants using transplants and nursery plugs
	Prevent/ encourage leaf litter accumulation	Remove leaf litter to enhance germination
		Encourage leaf litter development in new planting
	Increase soil fertility	Use fertilizer
		Add lime to stabilize the soil
		Add organic matter
		Use soil scarification or ploughing to enhance germination
		Use soil disturbance to enhance germination (excluding scarification or ploughing)
		Enhance soil compaction
	Reduce soil fertility	
Actions to improve survival and growth rate of planted trees	Replant trees	Fence to prevent grazing after tree planting
		Use prescribed fire after tree planting
		Mechanically remove understory vegetation after tree planting
		Manage woody debris before tree planting
		Add organic matter after tree planting
		Add lime to the soil after tree planting
		Use fertilizer after tree planting
		Use mechanical thinning before or after planting
		Use herbicide after tree planting
		Prepare the ground before tree planting
		Use different planting or seeding methods
		Cover the ground with straw after tree planting
		Use weed mats to protect planted trees
		Use tree guards or shelters to protect planted trees
		Use shading for planted trees
		Infect tree seedlings with mycorrhizae
		Introduce leaf litter to forest stands
		Transplant trees
		Use pioneer plants or crops as nurse-plants
		Reduce erosion to increase seedling survival
		Apply insecticide to protect seedlings from

		invertebrates
		Apply fungicide to protect seedlings from fungal diseases
		Improve soil quality after tree planting (excluding applying fertilizer)
		Water seedlings
		Plant a mixture of tree species to enhance the survival and growth of planted trees
Education and awareness raising		Raise awareness amongst the general public through campaigns and public information
		Provide education programmes about forests

Forest Conservation is part of the series of Synopses of Conservation Evidence, which are linked to the online resource www.ConservationEvidence.com.

This synopsis is part of the Conservation Evidence project and provides a useful resource for conservationists. It forms part of a series designed to promote a more evidence-based approach to biodiversity conservation. Others in the series include *Amphibian, Bat, Bee, Bird* and *Farmland Conservation* and many others are in preparation. The aim is to cover different species groups and habitats, gradually building into a comprehensive summary of evidence on the effects of conservation interventions for all global biodiversity.

This book brings together and summarises the available scientific evidence and experience relevant to the practical conservation of forest habitat.

The authors consulted an international group of forest experts and conservationists to produce a thorough summary of what is known, or not known, about the effectiveness of forest conservation actions across the world.

www.ConservationEvidence.com