Peatland Conservation

Global evidence for the effects of interventions to conserve peatland vegetation

Nigel G. Taylor, Patrick Grillas & William J. Sutherland

SYNOPSES OF CONSERVATION EVIDENCE SERIES

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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 5,400 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 of possible interventions. 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 many are 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 5,400 pieces of evidence, all available in a searchable database on the website www.conservationevidence.com.

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

Of the **125** peatland conservation interventions in this synopsis, none have been the subject of a specific systematic review. We extracted references from three systematic reviews on topics related to this synopsis: two which included studies from wet heaths in addition to peatlands (Gleaves *et al.* 2013; Stewart *et al.* 2014), and one whose focus was the effect of peat depth on restoration interventions (Lindsay & Clough 2016).

We feel that it would be useful, and possible given the amount of evidence we captured, to systematically review the effects of rewetting on peatland vegetation.

Scope of the Peatland Conservation synopsis

This synopsis reports the effects of interventions to conserve, restore or create peatland vegetation. We define peatlands as areas with wet peaty soils[1]. Peat is poorly decomposed organic matter, formed from dead plants in water-saturated environments. Studies in this synopsis may test interventions to conserve whole peatland plant communities or to conserve selected defining species within those communities.

The main vegetation types covered in the synopsis are bogs, fens, fen meadows and tropical peat swamps (see Table 1 for further descriptions). We include studies of fen meadows, as long as they are on wet peaty soils, because they share many species with fens and might be the only realistic target for restoration of severely degraded fens. Studies that restore or create peat-forming vegetation on non-peat soils are also within the scope of the synopsis, because these areas might develop into peatlands in the future. Non-English names for peatlands within the scope of the synopsis include *tourbières, bas marais* (French), *turberas*, *bofedales, pomponales* (Spanish), *hochmooren*, *niedermooren* (German), *turvemaat* (Finnish), *torvmarken* (Swedish) and *torfjaniky* (Russian).

This synopsis does not include information about conserving shrub-dominated wet heathlands, reed-dominated reedbeds, mangroves or salt marshes (even if these are on

[1] Peatlands are sometimes defined, more widely, as any area with a naturally accumulated layer of peat at the surface (e.g. Joosten & Clarke 2002). This synopsis focuses on vegetation typical of wet peatlands only.

wet peaty soils), vegetation on dry peat soils (e.g. dry heathland, forests on permanently drained peat) or wetland vegetation on non-peat soils (e.g. marshes, non-peat swamps).

Evidence from all around the world is included. Peatlands cover around 1–3% of the world's surface (depending on how they are defined) with particularly large areas in Canada, Russia, Northern Europe, South East Asia and the Congo Basin (Joosten & Clarke 2002; Dargie *et al.* 2017; Xu *et al.* 2018). Any apparent bias in this synopsis towards evidence from some regions reflects biases in the known distribution of peatlands and available published research.

Because of the wide scope of this synopsis, both in vegetation types and geographically, we encourage you to pay particular attention to the setting of each study when considering the evidence. You may need to consult the original references to get a full understanding of the study system and its history. Remember that the same change in vegetation might be desirable in one type of peatland, or one location, but undesirable in another.

Table 2 summarises the metrics used in this synopsis. The key metrics quantify the overall composition (which species or plant groups are present, and their abundance or cover) or physical structure (height, biomass, density) of peatland vegetation. We predefine some key plant species for which we have tried to consistently report data, whether they were affected by an intervention or not. Additionally, we report data for any species or groups that were important in a particular study e.g. dominant/characteristic species, or plant groups which showed large responses to the intervention. Finally, we use some additional metrics for interventions where effects on vegetation are rarely reported (e.g. habitat protection, campaigning, education and awareness-raising interventions). These metrics are intermediate outcomes that may contribute to habitat conservation.

Topics which fall outside the scope of this synopsis include:

- Studies in artificial environments such as laboratories, greenhouses and mesocosms (except those that test interventions to complement planting or aid planted vegetation, in a form that would be used in the field; see Chapter 13).
- Interventions to conserve rare plants (that exist in few locations or are not abundant where they do occur).
- Organisms other than plants (such as birds or amphibians: these are covered in other Conservation Evidence synopses and on www.conservationevidence.com).
- Effects of interventions on ecosystem functions (e.g. peat formation) and services (e.g. carbon storage), although these often benefit if vegetation is conserved.
- Effects of interventions on genetic diversity (which is sometimes a goal of conservation, especially for rare species or species used by humans).
- A detailed assessment of the evidence for "doing nothing" to conserve peatland vegetation, either in pristine/natural peatlands (where performing no active intervention may be the best option to conserve vegetation) or in degraded peatlands (where performing no active intervention is an option).

Table 1 Description of the main vegetation types covered in the Peatland Conservation synopsis. Circled letters are used throughout the synopsis to indicate the vegetation type to which each intervention is relevant. Key terminology is in bold italics.

	Description	Typical Vegetation
Bogs Ⓑ	Water and nutrients mainly from rain. Acidic. Low in nutrients. Most common in temperate or boreal regions. Two main types are ***raised bogs*** (on a dome of peat) and ***blanket bogs*** (draped across landscapes). Create peat when in good condition.	• Mosses e.g. bog mosses *Sphagnum* spp. • Herbs e.g. cottongrasses *Eriophorum* spp., grasses *Calamagrostis* spp. and *Molinia* spp. Sometimes rushes and restiads. • Small shrubs e.g. heather *Calluna vulgaris*, cross-leaved heath *Erica tetralix*, crowberry *Empetrum nigrum*, *Vaccinium* spp. • Sometimes trees e.g. alder *Alnus* spp., ash *Fraxinus* spp., spruce *Picea* spp. We use the term ***forested bogs*** to describe bogs with natural/desirable tree cover.
Fens Ⓕ	Water and nutrients from ground water as well as rain. More nutrients and less acidic than bogs, but variable. We sometimes distinguish ***poor fens*** (with lower pH, more similar to bogs) and ***rich fens*** (with higher pH). Not intensively managed, but may be harvested. Create peat when in good condition.	• Herbs e.g. sedges *Carex* spp., *Cladium* spp. and *Schoenus* spp., rushes *Juncus* spp., some common reed *Phragmites australis* • Mosses e.g. *Scorpidium* spp., *Calliergon* spp., *Warnstorfia* spp. • Sometimes trees/shrubs e.g. alder *Alnus* spp., ash *Fraxinus* spp., black spruce *Picea mariana*, Scots pine *Pinus sylvestris*. We use the terms ***forested fen*** or ***carr*** to describe fens with natural/desirable cover of woody plants.
Fen meadows Ⓕ	Derived from fens, but slightly drained and maintained by regular management such as mowing or grazing. Not currently forming peat, but based on peat or peat soils. Water table still high enough to influence vegetation.	• Herbs e.g. sedges *Carex* spp. and *Cladium* spp., purple moor grass *Molinia caerulea*, thistles *Cirsium* spp. Fewer tall reeds and rushes than fens. • Mosses similar to those in fens • No trees or shrubs
Tropical peat swamps Ⓢ	Forests on peat domes in the tropics. Water source is rainfall. Strictly a type of bog, but with distinct ecology and some unique conservation challenges compared to temperate/boreal bogs. Form peat when in good condition.	• Trees e.g. jelutong *Dyera polyphylla*, and balangeran *Shorea balangeran* in South East Asia; palms e.g. moriche *Mauritia flexuosa* in South America. • Some herbs/shrubs growing on ground or trees e.g. pandans *Pandanus* spp., sedges *Thorachostachyum bancanum*, pitcher plants *Nepenthes* spp.

Table 2 Summary of metrics used in the Peatland Conservation synopsis. The key metrics are used for all interventions. The additional metrics are used for some interventions in Chapter 14 (Habitat Protection) and Chapter 15 (Education and Awareness).

Key Metrics	
Plant community	• Plant community composition (Combination of which species are present and their relative cover/abundance. Summarized for the overall community. Often presented as a graphical analysis.) • Overall plant species richness/diversity[1]
Characteristic vegetation[2]	• Richness/diversity of 'peatland- or wetland-characteristic' plants • Cover/abundance of 'peatland- or wetland-characteristic' plants
Vegetation abundance	• Cover/abundance of herbaceous plants (forbs, grasses, sedges, reeds and rushes). Where possible, results reported separately for these groups and key taxa: true sedges *Carex* spp., cottongrasses *Eriophorum* spp., purple moor grass *Molinia caerulea* • Cover/abundance of mosses or bryophytes (mosses, liverworts and hornworts combined). Where possible, results reported separately for bog mosses *Sphagnum* spp. • Cover/abundance of trees/shrubs • Overall vegetation cover[3]
Vegetation structure	• Vegetation biomass (always above-ground, usually dry mass) • Vegetation height • Vegetation density
Individual plants[4]	• Germination • Survival • Growth

Additional Metrics	
Peatland habitat	• Peatland protection, restoration or creation (when this is the outcome of an intervention, not the intervention itself; usually reported as an area)
Behaviour change	• Change in threat (e.g. reduced logging activity) • Change in desirable behaviour (e.g. uptake of sustainable farming practices; change in purchasing behaviour; change in skills)
Knowledge change	• Change in understanding, awareness or attitude (e.g. increased awareness of the value of peatlands; increased knowledge of management techniques; intention to change behaviour)

(1) Some peatlands naturally contain few plant species. An increase in species richness might indicate degradation. Consider which species are present as well as how many there are. We have tried to report responses of key peatland species, or peatland-characteristic species overall, where clearly reported in original papers.

(2) Groups of plants that a study describes as characteristic of their focal peatland e.g. "bog-characteristic species", "target fen species" or "wetland indicator species".

(3) Cover often exceeds 100% because it can include overlapping layers of vegetation. For example, a lawn of grass with 100% cover might be overgrown by leaves of a tree with 20% cover, giving 120% total cover. As for species richness, also consider which species are present. High vegetation cover of non-native or non-peatland species might not be desirable (although it could stabilize and protect peat in the short term). We have tried to report further information about cover of plant species or types, where clearly reported in original papers.

(4) Typically reported as a response of planted/introduced vegetation (Chapters 12 and 13).

How we decided which conservation interventions to include

A list of **125** peatland conservation interventions was developed and agreed in partnership with an Advisory Board made up of international conservationists and academics with expertise in peatland conservation. We have tried to include all actions that have been carried out or advised to benefit peatland vegetation. Please note that:

- Inclusion of an intervention is not an endorsement or an indication that it is effective. For example, we have included many interventions for which there is no evidence for their effects.
- Many of the interventions are suitable for specific peatland types or in specific contexts. Interventions that might be beneficial for fens could be very damaging to bogs. Some interventions might only be effective if combined with another (e.g. drained peatlands might need to be rewetted before sowing mosses). The background sections, main text summarizing each study, and even the original references should be consulted to fully understand the context of each study.
- Active intervention may not the best option to conserve a peatland. On relatively undisturbed peatlands the best action might be no action at all, or action to protect rather than restore (see Chapter 14).
- Most of the listed interventions are reactive (treating the effects of threats e.g. cutting down forestry plantations). This is not meant to discourage proactive conservation (addressing root causes of threats). Many proactive interventions are simply beyond the scope of a peatland-focused synopsis (e.g. interventions to tackle climate change in general).

The listed interventions are often broader in scope than the summarized evidence. We searched for evidence of the effects of each intervention on vegetation in all peatland types where you might want to carry out the intervention. The summarized evidence reflects the evidence we captured, not the intended scope of the intervention.

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

How we reviewed the literature

This synopsis includes studies of conservation interventions for peatland vegetation, published in 2016 or earlier, from the following sources:

- Systematic searches of over 230 ecology and conservation journals and reports (see Appendix 1). Most of these have been searched as part of the Conservation Evidence project and relevant papers stored in a central database. Twelve specialist wetland/peatland/botanical journals were searched specifically for the Peatland Conservation synopsis. Journals and reports were generally searched from their first issue to the end of 2016.
- Other publications in the Conservation Evidence database relevant to this synopsis (e.g. those recommended by advisory boards for previous synopses).
- Other publications recommended by the Advisory Board for this synopsis.

- Other publications identified during the summarizing process for this synopsis (e.g. from within reviews or systematic reviews).

The criteria for inclusion of studies were 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, we excluded predictive modelling studies and correlative studies of relationships between peatland vegetation and environmental characteristics without a clear link to an intervention.

Evidence written in any language was included when it was identified, although most of the sources searched were in English.

Altogether **296 studies** from **162 papers/reports** were identified and summarized. In this synopsis, a study is a conceptually distinct test of an intervention (e.g. performed in a different place, at a different time, with a different method, reporting different results and/or analyzed separately). Thus, one paper can contain multiple studies. Each study is reported in a separate synopsis paragraph.

How the evidence is summarized: classification by threats

The conservation interventions are grouped primarily according to the threat they directly address. Threats are as defined in the IUCN Unified Classification of Direct Threats (http://www.iucnredlist.org/technical-documents/classification-schemes/threats-classification-scheme). Not all IUCN threat types are included: only those that threaten peatlands and for which realistic conservation interventions have been suggested.

In most cases, it is clear which main threat a particular intervention is meant to alleviate or counteract. However some 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. For these interventions, we created additional chapters corresponding to the IUCN Classification of Conservation Actions (http://www.iucnredlist.org/technical-documents/classification-schemes/conservation-actions-classification-scheme-ver2) as follows:

Chapter	IUCN Conservation Action
12: Habitat creation and restoration	Land/water management
13: Actions to complement planting	Land/water management
14: Habitat protection	Land/water protection
15: Education and awareness	Education and awareness

Normally, each intervention is listed in only one place: under the threat it addresses. Similar interventions might be listed in more than one place when they address slightly different threats e.g. mowing as part of a traditional management regime (Chapter 8) and mowing to control problematic plant species where there is no traditional management regime (Chapter 9). In these cases, there is clear cross-referencing between interventions.

Normally, each study is listed in only one place: under the intervention it tests. Studies might be listed in more than one place if they carry out multiple interventions at once and the effects of these interventions cannot be separated. This is clearly noted within each summary paragraph and within the key messages.

How the evidence is summarized: presentation

At the start of each chapter, a series of key messages provides a rapid overview of the evidence. These messages are condensed from the summary bullet points for each intervention. Then, evidence for the effects of each intervention is presented as follows:

- Bullet points summarize the evidence for each intervention, categorized by outcome metrics (see Table 2). Each bullet point gives the total number of studies reporting each metric and summarizes what those studies found. These sections focus on the most commonly-reported metrics for each intervention, whilst always referencing all studies that tested the intervention.
- A Background box explains the context of each intervention to help you interpret the evidence. CAUTION indicates potential undesirable effects of the intervention on any aspect of the environment. Related interventions are cross-referenced. References are given for each background section.
- The main text presents studies in chronological order i.e. the most recently published evidence is presented at the end. Numbered References are provided for each intervention. Under each intervention, paragraphs sharing the same reference number (e.g. 1a, 1b, 1c) are all from the single paper/report with that number.
- A key gives a rapid overview of the broad peatland types to which the intervention is most relevant (circled letters Ⓑ) and for which we captured evidence (bold, dark letters **Ⓑ**). The intervention is less relevant, or not relevant, to peatland types that are not circled (B). B = bogs; F = fens and fen meadows; S = tropical peat swamps.

For many interventions, there are no studies summarized and we state that "*We captured no evidence for the effect of the intervention*". This means we did not identify any studies that directly tested the intervention and quantitatively reported the effects. It does not mean that we only identified studies that found no effect: these would be summarized in the synopsis.

Often, papers contain multiple studies testing different interventions. Each study is summarized in its own paragraph under the relevant intervention (e.g. van Duren *et al*. 1998). Sometimes, papers contain multiple conceptually distinct studies testing one intervention. Each study is summarized in a separate paragraph, but linked to the single paper (e.g. referenced as 6a, 6b, 6c). Sometimes studies from different papers use the same experimental set-up to test the same intervention. All are summarized individually if they present at least partially different results, but the shared experimental set-up is clearly indicated (e.g. Section 12.2). If different papers present identical results from the same experimental set-up, the results are only summarized once but all papers are provided as references (e.g. Section 3.4).

The information in this synopsis is available (a) as a pdf, free to download from www.conservationevidence.com and (b) 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 study clear. Table 3 below defines the terms that we have used to do this.

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

Table 3 Terminology used to describe evidence in Conservation Evidence synopses

Term	Meaning
Replicated	The intervention was repeated on more than one individual or site. In conservation and ecology, the number of replicates is much smaller than it would be for medical trials (when thousands of individuals are often tested). If the replicates are sites, pragmatism dictates that between five and ten replicates is a reasonable amount of replication, although more would be preferable. We provide the number of replicates wherever possible. In the case of planting or vegetation introduction, replicates should be sites, not individuals.
Randomized	The intervention was allocated randomly to individuals or sites. This means that the initial condition of those given the intervention is less likely to bias the outcome.
Paired	Sites or plots are considered in pairs, when one was treated with the intervention and the other was not. Pairs or blocks 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.
Controlled	Individuals or sites treated with the intervention are compared with designated control individuals or sites not treated with the intervention.
Before-and-after	Monitoring of effects was carried out before and after the intervention was imposed.
Site comparison	A study that considers the effects of interventions by comparing sites that have historically had different interventions or levels of intervention.
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. Many environmental systematic reviews are available at: www.environmentalevidence.org.
Study	If none of the above apply, for example a study that has measured change in only one site and only after an intervention.

Taxonomy

We have not updated or standardized taxonomy. We report scientific names as reported in each reference, but have tried to use common names consistently throughout the synopsis. Where possible, common names and scientific names are both given the first time a species is mentioned in each paragraph. We use only common names in the key messages and bullet point summaries.

Significant results

Throughout the synopsis we have quoted results from papers/reports. Differences are significant and based on statistical hypothesis tests, unless specifically indicated with a sentence such as "*These results were not tested for statistical significance*" or "*No statistical tests were carried out*". Sometimes there are statistical tests in the original paper, but they do not reflect the results or the part of the study summarized in this synopsis. We use the word "*found*" to describe results that have (mostly) been tested for significance and "*reported*" to describe results that (mostly) have not.

How you can help to change conservation practice

If you know of evidence relating to peatland conservation that is not included in this synopsis, we invite you to contact us, via our website www.conservationevidence.com. If you have new, unpublished evidence, you can submit a paper to the *Conservation Evidence* journal. We welcome all papers reporting the effects of conservation interventions, whether the intervention worked as planned or not. We particularly welcome papers submitted by conservation practitioners.

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2. Threat: Residential and commercial development

Background

This chapter addresses damage to peatlands from developments with a large footprint, like residential areas, factories, shopping malls, golf courses and airports. Examples include the suburbs of Ushuaia in southern Argentina, built on drained and filled peatlands (de la Balze *et al.* 2004) and peatlands in Bulgaria threatened by construction of hotels, weekend houses and petrol stations (Hájek *et al.* 2010)

Related threats: construction of transportation and service corridors (Chapter 5); human intrusions and disturbance (Chapter 7); drainage (Chapter 8); pollution (Chapter 10). Related interventions: general habitat creation and restoration (Chapter 12); protection of peatlands from threats including development (Chapter 14).

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Key messages

2.1 *Remove residential or commercial development from peatlands* **0 studies**

We captured no evidence for the effect on peatland vegetation of removing residential or commercial development from peatlands.

2.2 *Retain/create habitat corridors in developed areas* **0 studies**

We captured no evidence for the effect on peatland vegetation, in habitat patches or within corridors, of retaining or creating habitat corridors in developed areas.

Interventions

2.1 Remove residential or commercial development from peatlands

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of removing residential or commercial development from peatlands.

Background

Removing urban areas, industrial facilities or tourist sites that have been built on or near peatlands could allow the vegetation to recover.

Related interventions: rewetting, because peatlands are often drained to allow development or are dried out by drainage of sites nearby (Section 8.1); habitat creation and restoration for formerly developed land (Chapter 12).

2.2 Retain/create habitat corridors in developed areas Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation, in habitat patches or within corridors, of retaining or creating habitat corridors in developed areas.

Background

Habitat corridors are strips that link two larger habitat patches – in this case preventing peatland patches being separated by development. By connecting the habitat patches, corridors could improve survival prospects of peatland plant populations. Seeds, pollen or vegetation fragments can be moved along corridors (e.g. by animals), maintaining populations and diversity in each patch (Damschen *et al.* 2006). CAUTION: Habitat corridors can also have negative effects, like allowing diseases, non-native species and fire to spread between patches (Resasco *et al.* 2014).

Related interventions: rewetting, because peatlands may be drained to allow development or are dried out by drainage of sites nearby (Section 8.1); habitat creation and restoration (Chapter 12).

Damschen E.I., Haddad N.M., Orrock J.L., Tewksbury J.J. & Levey D.J. (2006) Corridors increase plant species richness at large scales. *Science*, 313, 1284–1286.

Resasco J., Haddad N.M., Orrock J.L., Shoemaker D., Brudvig L., Damschen E.I., Tewksbury J.J. & Levy D.J. (1991) *Peat resources use in Canada: a national conservation issue*. Proceedings, International Peat Symposium. Duluth.

3. Threat: Agriculture and aquaculture

Background

This chapter addresses the threat from agriculture on peatlands. Large areas of peatland have been (and continue to be) converted to crop fields, livestock pasture or forest plantations: 75% of peatlands in Waikato, New Zealand (Environment Waikato 1998), over 85% in Germany and the Netherlands, and 15% in Canada (Strack 2008). Rice farms have replaced much of the peatland in Japan (Fujita et al. 2009). Palm oil, rubber and pulp wood plantations, along with smallholder farms, now cover around 50% of peatlands in Malaysia, Sumatra and Borneo (Miettinen *et al.* 2016).

Crop land and plantations contain unnatural communities of plants and animals. Grazing livestock consume and trample peatland vegetation, altering the community composition and vegetation cover. Grazed peatlands are also susceptible to erosion (e.g. Salvador *et al.* 2014). Farmed peatlands are often dry, having been deliberately drained (e.g. the Indonesian Mega Rice Project) or dried out by crops taking up water. Dry peatlands are more prone to burning. Drainage also affects water levels in neighbouring peatlands. However, agricultural peatlands can sometimes become unnaturally wet, for example when they sink under the weight of forest plantations and then flood. Interventions to address these related threats are considered in Sections 8.1 (drainage), 8.2 (flooding), 8.10, 8.11, 8.12 and 15.1 (fire).

Former agricultural land could be left to recover without any active intervention. Passive recovery is cheaper than active intervention, but may be ineffective. We do not assess the evidence for this in detail, but note that passive recovery can be slow and lead to communities dominated by weedy species, remnant crop species, or the few species that can most easily disperse (Blackham *et al.* 2014; Bart *et al.* 2015).

Related threats: biological resource use i.e. harvesting existing vegetation (Chapter 6); natural system modifications, such as drainage and changes to disturbance regimes, to make peatlands suitable for agriculture (Chapter 8); pollution from agriculture on or near to peatlands (Chapter 10). Related interventions: general habitat creation and restoration (Chapter 12); voluntary codes and payment schemes to protect peatlands (Chapter 14); education/training of landowners (Chapter 15).

Bart D. & Davenport T. (2015) The influence of legacy impacted seed banks on vegetation recovery in a post-agricultural fen complex. *Wetlands Ecology and Management*, 23, 405–418.

Blackham G.V., Webb E.L. & Corlett R.T. (2014) Natural regeneration in a degraded tropical peatland, Central Kalimantan, Indonesia: implications for forest restoration. *Forest Ecology and Management*, 324, 8–15.

Environment Waikato (1988) *Waikato State of the Environment Report.* Environment Waikato, Hamilton, New Zealand.

Fujita H., Igarashi Y., Hotes S., Takada M., Inoue T. & Kaneko M. (2009) An inventory of the mires of Hokkaido, Japan – their development, classification, decline and conservation. *Plant Ecology*, 200, 9–36.

Miettinen J., Shi C. & Liew S.C. (2016) Land cover distribution in the peatlands of Peninsular Malaysia, Sumatra and Borneo in 2015 with changes since 1990. *Global Ecology and Conservation*, 6, 67–78.

Salvador F., Monerris J. & Rochefort L. (2014) Peatlands of the Peruvian Puna ecoregion: types, characteristics and disturbance. *Mires and Peat*, 15, Article 3.

Strack M. (ed.) (2008) *Peatlands and Climate Change.* International Peat Society, Finland.

Key messages

Multiple farming systems

3.1 *Implement 'mosaic management' of agriculture* **0 studies**

We captured no evidence for the effect on peatland vegetation of implementing mosaic management in agricultural systems.

3.2 *Retain/create habitat corridors in farmed areas* **1 study**

Vegetation structure: One study in Indonesia found that a peat swamp forest corridor contained 5,819 trees/ha: 331 large trees, 1,360 saplings and 4,128 seedlings.

Overall plant richness/diversity: The same study recorded 18–29 tree species (depending on the size class) in the peat swamp forest corridor.

Wood and pulp plantations

3.3 *Cut/remove/thin forest plantations* **4 studies**

Herb cover: Three replicated studies (two also paired and controlled) in bogs in the UK and fens in Sweden reported that tree removal increased cover of some herbs including cottongrasses and sedges. One of the studies reported no effect on other herb species.

Moss cover: Two replicated studies, in bogs in the UK and a drained rich fen in Sweden, reported that tree removal reduced moss cover (fen-characteristic mosses or *Sphagnum* moss). However, one replicated, paired, controlled study in partly rewetted rich fens in Sweden reported that tree removal increased *Sphagnum* moss cover after eight years.

Overall plant richness/diversity: Two replicated, paired, controlled studies in rich fens in Sweden reported that tree removal increased total plant species richness, especially in rewetted plots.

3.4 *Cut/remove/thin forest plantations and rewet peat* **11 studies**

Plant community composition: Of three replicated studies in fens in Finland and Sweden, two found that removing trees/rewetting had no effect on the overall plant community composition and the other reported only a small effect. Two site comparison studies, in bogs and fens in Finland, found that removing trees/rewetting changed the community composition. It became less like forested and drained sites.

Characteristic plants: Two before-and-after studies (one site comparison, one controlled) in bogs and fens in Finland and Sweden reported that removing trees/rewetting increased the abundance of wetland-characteristic plants.

Moss cover: Five studies (four replicated, three site comparisons) in Sweden and Finland examined the effect of removing trees/rewetting on *Sphagnum* moss cover. Of these, two studies in bogs and fens found that removing trees/rewetting increased *Sphagnum* cover. One study in forested fens found no effect. Two studies in a bog and a fen found mixed effects amongst sites or species. Four studies (three replicated, two paired) in the UK and Finland examined the effect of removing trees/rewetting on cover of other mosses. Of these, three studies found that removing trees/rewetting reduced moss cover, but one study in forested fens found no effect.

Herb cover: Seven studies (two replicated, paired, controlled) in bogs and fens in the UK, Finland and Sweden reported that removing trees/rewetting increased cover of at least one group of herbs, including cottongrasses in four of five studies and sedges in three of three studies. One study reported that removing trees/rewetting reduced cover of purple moor grass.

Vegetation structure: Two replicated studies examined the effect of removing trees/rewetting on vegetation height. Of these, one study in a bog in the UK found that removing trees/rewetting increased ground vegetation height, but one study in a fen in Sweden reported no effect on canopy height after eight years. Two replicated, paired, site comparison studies in bogs and fens in Finland reported that thinning trees/rewetting reduced the number of tall trees present for 1–3 years after intervention (but not to the level of natural peatlands).

Overall plant richness/diversity: Of four replicated studies in fens in Sweden and Finland, two (also paired and controlled) reported that removing trees/rewetting increased plant species richness. The other two studies found no effect on plant species richness or diversity.

Livestock farming and ranching

3.5 ***Use barriers to keep livestock off ungrazed peatlands*** **0 studies**

We captured no evidence for the effect on peatland vegetation of using barriers to keep livestock off peatlands that have never (or not recently) been grazed.

3.6 ***Exclude or remove livestock from degraded peatlands*** **10 studies**

Plant community composition: Of two replicated, paired, controlled studies in bogs in the UK, one found that excluding sheep had no effect on the plant community. The other found that excluding sheep only affected the community in drier areas of the bog, favouring plants of dry moorlands.

Herb cover: Seven studies (six replicated, paired, controlled) in bogs and fens in the UK, Australia and the USA found that excluding/removing livestock had no effect on cover of key herb groups: cottongrass in five of five studies and sedge in two of two studies. However, one before-and-after study in a poor fen in Spain reported that rush cover increased after cattle were excluded, along with other interventions. One site comparison study in Chile found that excluding livestock, along with other interventions, increased overall herb cover but one replicated, paired, controlled study in bogs in Australia found no effect.

Moss cover: Five replicated, paired, controlled studies in bogs in the UK and Australia found that excluding livestock typically had no effect on *Sphagnum* moss cover. Three of the studies in the UK also found no effect on cover of other mosses. One before-and-after study in a poor fen in Spain reported that *Sphagnum* moss appeared after excluding cattle (and rewetting).

Tree/shrub cover: Five replicated, paired, controlled studies in bogs in the UK and Australia found that excluding livestock typically had no effect on shrub cover (heather or a heathland community). However, one of these studies found that excluding sheep increased heather cover in drier areas. Three studies (two site comparisons) in bogs in the UK, fens in the USA and a peatland in Chile found that excluding/removing livestock increased shrub cover.

Vegetation structure: One replicated, paired, controlled study in a bog in the UK found that excluding sheep increased total vegetation, shrub and bryophyte biomass but had no effect on biomass of grass-like herbs.

3.7 ***Reduce intensity of livestock grazing*** **1 study**

Vegetation cover: One replicated, paired, controlled study in bogs in the UK found greater cover of total vegetation, shrubs and one of two cottongrass species under lower grazing intensities.

Vegetation structure: The same study found that vegetation biomass was higher under lower grazing intensities.

3.8 ***Change type of livestock*** **0 studies**

We captured no evidence for the effect on peatland vegetation of changing livestock type.

3.9 ***Change season/timing of livestock grazing*** **0 studies**

We captured no evidence for the effect on peatland vegetation of changing the season or timing of livestock grazing.

Interventions: Multiple farming systems

3.1 Implement 'mosaic management' of agriculture

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- We captured no evidence for the effect on peatland vegetation of implementing mosaic management in agricultural systems.

Background

Mosaic management involves managing neighbouring patches of land in different ways. For example, patches of agricultural land could be interspersed with natural vegetation that is never harvested. *Sphagnum* mosses could be farmed in patches on bogs; reeds farmed in patches of fens; trees producing latex, fruit or medicines farmed in patches of tropical peat swamps (e.g. Giesen 2015). There is some evidence that mosaic farmland management may benefit wildlife such as birds (Dicks *et al.* 2013).

Note that mosaic management of peatlands is only possible if the farmed areas are kept wet. Draining patches of peatland for agriculture will lower the entire local water table, affecting land beyond the focal agricultural area.

Related intervention: implement mosaic management when harvesting wild biological resources (Section 6.5).

Dicks L.V., Ashpole J.E., Dänhardt J., James K., Jönsson A., Randall N., Showler D.A., Smith R.K., Turpie S., Williams D. & Sutherland W.J. (2013) *Farmland Conservation: Evidence for the Effects of Interventions in Northern and Western Europe*. Pelagic Publishing, Exeter.

Giesen W. (2015) Utilising non-timber forest products to conserve Indonesia's peat swamp forests and reduce carbon emissions. *Indonesian Journal of Natural History*, 3, 10–19.

3.2 Retain/create habitat corridors in farmed areas

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- **One study** examined the effect on peatland vegetation, in habitat patches or within corridors, of retaining or creating habitat corridors in farmed areas. This study was in a tropical peat swamp.
- **Vegetation structure (1 study):** One study in Indonesia[1] found that a peat swamp forest corridor contained 5,819 trees/ha. This included 331 large trees/ha, 1,360 saplings/ha and 4,128 seedlings/ha.
- **Overall plant richness/diversity (1 study):** The same study[1] recorded 18–29 tree species in the peat swamp forest corridor (the number of species depending on the size class).

Background

Habitat corridors are strips that link two larger habitat patches – in this case preventing peatland patches from being separated by agricultural land. By connecting the habitat patches, corridors could improve survival prospects of peatland plant populations. Seeds, pollen or vegetation fragments can be moved along corridors (e.g. by animals), maintaining populations and diversity in each patch (Damschen *et al.* 2006). CAUTION: Habitat corridors can have negative effects. For example, corridors can allow diseases, non-native species and fire to spread between patches (Resasco *et al.* 2014).

To help you interpret the summarized study, peat swamps in South East Asia typically contain 30–122 large (trunk diameter >10 cm) tree species/ha (Posa *et al*. 2011).

Related interventions: retain/create habitat corridors in areas of energy production or mining (Section 4.2).

Damschen E.I., Haddad N.M., Orrock J.L., Tewksbury J.J. & Levey D.J. (2006) Corridors increase plant species richness at large scales. *Science*, 313, 1284–1286.

Posa M.R.C., Wijedasa L. & Corlett R.T. (2011) Biodiversity and conservation of tropical peat swamp forests. *BioScience*, 61, 49–57.

Resasco J., Haddad N.M., Orrock J.L., Shoemaker D., Brudvig L., Damschen E.I., Tewksbury J.J. & Levy D.J. (2014) Landscape corridors can increase invasion by an exotic species and reduce diversity of native species. *Ecology*, 95, 2033–2039.

A study in 2007 in a peat swamp forest corridor in Indonesia (1) reported that the corridor contained 5,819 trees/ha (of all sizes) and 18–29 species (depending on size class). There were 331 large trees/ha, 1,360 saplings/ha and 4,128 seedlings/ha. There were 27 different species of large tree, 18 species of sapling and 29 species of seedling (total number of species not reported). The tallest trees were 48 m high. The study does not report comparable data for natural peat swamp forests. In 2007, one 100 x 100 m plot was established in a forest corridor (100–500 m wide), retained for nature conservation within a red wattle *Acacia crassicarpa* plantation. The water table was approximately 1 m lower than in natural peat swamp forest. Trees at all life stages were counted, measured and identified: large trees (trunk diameter >10 cm) in the entire plot, saplings (diameter 5–10 cm) in twenty-five 5 x 5 m subplots, and seedlings (diameter <5 cm) in twenty-five 2 x 2 m subplots.

(1) Gunawan, H., Page, S.E., Muhammad, A., Qomar, N., Helentina, T., Hakim, A., Yanti, M.M. & Darmasanti, P. (2007) *Peat swamp forest regeneration using green belts in a timber estate in Riau, Sumatra Indonesia*. Proceedings of the International Symposium and Workshop on Tropical Peatland, 27–29 August 2007, Yogyakarta, Indonesia, 83–88.

Interventions: Wood and pulp plantations

3.3 Cut/remove/thin forest plantations Ⓑ Ⓕ S

- **Four studies** examined the effect on peatland vegetation of cutting/removing forest plantations: one in bogs[1] and three in fens[2,3,4]. The studies in the fens[2,3,4] were all based, at least in part, on the same experimental set-up.
- **Herb cover (3 studies):** Three replicated studies (two also paired and controlled) in bogs in the UK[1] and fens in Sweden[2,3] reported that tree removal increased cover of some herb species including cottongrasses[1,2] and sedges[3]. One of the studies[2] reported no effect of tree removal on other herb species.
- **Moss cover (3 studies):** One replicated, paired, controlled study in bogs in the UK[1] reported that tree removal reduced cover of forest-characteristic mosses. One replicated before-and-after study in a drained rich fen in Sweden[2] reported that *Sphagnum* moss cover decreased over three years following tree removal. However, one replicated, paired, controlled study in partly rewetted rich fens[3] reported that *Sphagnum* cover increased over eight years following tree removal.
- **Overall plant richness/diversity (2 studies):** Two replicated, paired, controlled studies in rich fens in Sweden[3,4] reported that tree removal increased total plant species richness. However, one of these studies[4] reported a much smaller effect of tree removal in rewetted plots than in drained plots.

Background

This section considers the threat from forest plantations on peatlands. Forests can dry out peatlands (trees take up water and reduce inputs from rainfall), create shade that prevents ground vegetation from growing, and cause peat to subside under the weight of the trees (Lindsay *et al.* 2014). Peatland vegetation may recover if trees are felled without any additional manipulation of the water table: it may rise on its own because water is no longer taken up by the trees, or as drainage ditches collapse and fill with debris.

This section includes cutting/thinning of *afforested* peatlands (where trees have been deliberately planted) and *tree-colonized* peatlands (where trees have colonized by themselves, for example after drainage for forestry). By comparison, peatlands with natural tree cover are described as *forested* or *swamps*. *Clear-cutting* refers to felling and removal of all trees from a site.

Related interventions: rewetting, because peatlands are often drained for forestry (Section 8.1); rewetting combined with tree removal (Section 3.4); cutting large trees/shrubs on peatlands, not within forestry plantations (Section 9.6).

Lindsay R., Birnie R. & Clough J. (2014) *Ecological Impacts of Forestry on Peatlands*. IUCN UK Peatland Programme Briefing Note No. 4.

A replicated, paired, controlled study in two afforested blanket bogs in Scotland, UK (1) reported that plots where trees were felled developed greater cover of sheathed cottongrass *Eriophorum vaginatum*, and typically less cover of forest mosses, than plots that remained forested. These results were not tested for statistical significance. After five years, felled plots had greater cottongrass cover (16–45%) than forested plots (11–19%). In contrast, felled plots typically had less cover of forest mosses: silk moss *Plagiothecum undulatum* in four of four comparisons (felled: <1%; forested: 3–6%) and plait moss *Hypnum cupressiforme* in three of four comparisons (felled: 18–35%; forested: 44–57%). Amongst felled plots, the effect on cottongrass and silk moss was generally larger when debris was left in place rather than removed. Between 1996 and 1998, six blocks of six 40 x 100 m plots were established in drained, conifer-forested bogs. Twelve plots (two plots/block) received each felling treatment: felling and removing debris, felling and leaving debris in place, or no felling. Within each treatment, half of the plots were also rewetted. Five years after intervention, vegetation cover was recorded (details not reported).

A replicated before-and-after study in 2002–2005 in two drained, tree-colonized, rich fens in Sweden (2) reported that following tree removal, there were small changes in plant community composition and cover. These results are not based on tests of statistical significance. The overall composition of the plant community changed over three years following tree removal (data reported as a graphical analysis). Cover was reported for the most abundant plant species. For example, in one fen, *Sphagnum* moss cover was 43% before tree removal but 28% three years after. Cover of common cottongrass *Eriophorum angustifolium* was <1% before but 5% after. Across both fens, cover remained relatively stable for purple moor grass *Molinia caerulea* (before: 55%; after: 50%), common reed *Phragmites australis* (4% before and after) and sedges *Carex* spp. (0–1% before and after). In late 2002, all trees were cut and removed from two drained 50 x 150 m plots (one plot/fen). Vegetation cover was estimated before (2002) and after (2005) tree removal in 4–16 quadrats (each 0.25 m^2) in the centre of each plot. This study was based on the same experimental set-up as (3) and (4).

A replicated, paired, controlled, before-and-after study in 2002–2010 in three tree-colonized rich fens in Sweden (3) found that following tree removal, there were increases in plant species richness and bryophyte, grass and sedge cover, but not cover of fen-characteristic plants. In cleared plots, plant species richness increased from 9 plant species/0.25 m^2 before tree removal to 11 species/0.25 m^2 eight years after, although it peaked at 12 species/0.25 m^2 after three years. Cover increased of *Sphagnum* mosses (from 10% before tree removal to 15% eight years after), wetland-characteristic bryophytes (from 27 to 37%), grasses (from 2 to 4%) and sedges (from 1 to 3%). There was no significant change in cover of fen-characteristic mosses or vascular plants (data not reported). In plots that remained forested, there was no change in species richness or vegetation cover. In winter 2002/2003, in each of three forested fens, trees were removed from one 50 x 300 m plot whilst an adjacent plot was left forested. Half of each plot remained drained whilst half was rewetted. Between 2002 (before intervention) and 2010, cover of every plant species was estimated at 40 points/plot, in 0.25 m^2 quadrats. This study was based on the same experimental set-up as (2) and (4).

A replicated, paired, controlled, site comparison study in 2002–2010 involving three tree-colonized rich fens in Sweden (4) reported that tree removal increased plant species richness, especially in drained plots, but found that it had no effect on the height of new vegetation. Amongst plots that remained drained, those that were clear-cut had higher plant species richness after eight years than those that remained forested (clear-cut: 13; forested: 9 species/0.25 m^2). Amongst rewetted plots, tree removal had a much smaller effect (clear-cut: 13; forested: 14 species/0.25 m^2). These results were not tested for statistical significance. Tree removal had no significant effect on canopy height (of new vegetation) in drained plots (clear-cut: 6 m; forested: 6 m) or rewetted plots (clear-cut: 5 m; forested: 5 m). For comparison, a nearby natural (undrained and unforested) fen contained 9 plant species/0.25 m^2 and had a canopy height of 1 m. These were significantly greater in the clear-cut plots. Around winter 2002/2003, trees were removed from one 50 x 300 m plot in each tree-colonized fen. An adjacent plot was left forested. Half of each plot was also rewetted whilst half remained drained. In 2010, plant species and canopy height were recorded at 40 points/plot, in 0.25 m^2 quadrats. The natural fen was sampled in 1978. This study was based on the same experimental set-up as (2) and (3).

(1) Anderson R. (2010) *Restoring afforested peat bogs: results of current research*. Forestry Commission Research Note 6.
(2) Mälson K., Sundberg S. & Rydin H. (2010) Peat disturbance, mowing, and ditch blocking as tools in rich fen restoration. *Restoration Ecology*, 18, 469–478.
(3) Hedberg P., Kotowski W., Saetre P., Mälson K., Rydin H. & Sundberg S. (2012) Vegetation recovery after multiple-site experimental fen restorations. *Biological Conservation*, 147, 60–67.
(4) Hedberg P., Saetre P., Sundberg S., Rydin H. & Kotowski W. (2013) A functional trait approach to fen restoration analysis. *Applied Vegetation Science*, 16, 658–666.

3.4 Cut/remove/thin forest plantations and rewet peat Ⓑ Ⓕ S

- **Eleven studies** examined the effect of cutting/removing trees and rewetting peat (in combination): six in fens[1,4,5,6,8,10], two in bogs[2,7], and three in both fens and bogs[3,9,11]. In four of the studies[5,9,10,11] the peatlands naturally contained some trees. Three studies[4,6,8] were based on one experimental set-up, and two studies[9,11] were based on another.

- **Plant community composition (5 studies):** Of three replicated studies in fens, two in Finland[5,10] found that removing trees/rewetting had no effect on the overall plant community composition whilst one in Sweden[4] reported only a small effect. Two site comparison studies in bogs and fens in Finland[3,9] found that removing trees/rewetting changed the overall plant community composition. It became less like sites that remained forested and drained.
- **Characteristic plants (2 studies):** Two before-and-after studies (one site comparison, one controlled) in bogs and fens in Finland[3] and Sweden[6] reported that removing trees/rewetting increased the abundance of wetland-characteristic plants.
- **Moss cover (6 studies):** Of five studies that examined the effect of removing trees/rewetting on *Sphagnum* moss, two replicated, paired studies in bogs and fens in Sweden[6] and Finland[9] found that the intervention increased *Sphagnum* cover. One replicated, before-and-after, site comparison study in forested fens in Finland[5] found no effect. Two before-and-after studies in a bog in Finland[3] and a fen in Sweden[4] found mixed effects depending on site[3] or species[4]. Additionally, three studies (two replicated and paired) in peatlands in the UK[2] and Finland[3,9] found that removing trees/rewetting reduced cover of non-*Sphagnum* or forest-characteristic mosses. However, one replicated, before-and-after, site comparison study in forested fens in Finland[5] found no effect of thinning trees/rewetting on forest mosses.
- **Herb cover (7 studies):** Seven studies (including two replicated, paired, controlled) in bogs and fens in the UK[2,7], Finland[1,3,5] and Sweden[4,6] reported that removing trees/rewetting increased cover of at least one group of herbs, including cottongrasses[1,2,3,5] and sedges[4,5,6]. However, one of these studies[4] reported loss of cottongrass from a fen where it was rare before intervention, along with reduced purple moor grass cover.
- **Vegetation structure (4 studies):** One replicated site comparison study in a bog in the UK[7] found that removing trees/rewetting increased ground vegetation height. One replicated, paired, controlled study in a fen in Sweden[8] reported that removing trees/rewetting had no effect on canopy height after eight years. Two replicated, paired, site comparison studies in bogs and fens in Finland[9,11] reported that thinning trees/rewetting reduced the number of tall trees present for 1–3 years (although not to the level of natural peatlands).
- **Overall plant richness/diversity (4 studies):** Two replicated, paired, controlled studies in rich fens in Sweden[6,8] reported that removing trees/rewetting increased plant species richness. However, two replicated studies in fens in Finland[5,10] found that removing trees/rewetting had no effect on total plant species richness or diversity.

Background

This section considers the threat from forest plantations on peatlands. Forests can dry out peatlands (trees take up water and reduce inputs from rainfall), create shade that prevents ground vegetation from growing, and cause peat to subside under the weight of the trees (Lindsay *et al.* 2014). Peatland vegetation may recover if trees are felled, but the process may be sped up by additional interventions to raise the water table to rewet the surface peat (e.g. blocking drainage ditches).

This section includes cutting/thinning, combined with rewetting, of *afforested* peatlands (where trees have been deliberately planted) and *tree-colonized* peatlands (where trees have colonized by themselves, for example after drainage for forestry). By comparison, peatlands with natural tree cover are described as *forested* or *swamps*. *Clear-cutting* refers to felling and removal of all trees from a site.

Related interventions: removal of forestry plantations alone (Section 3.3); rewetting alone (Section 8.1); cutting large trees/shrubs on peatlands, not within forestry plantations (Section 9.6).

Lindsay R., Birnie R. & Clough J. (2014) *Ecological Impacts of Forestry on Peatlands*. IUCN UK Peatland Programme Briefing Note No. 4.

A controlled, before-and-after study in 1994–1996 in a fen in Finland (1) reported that clear-cut and rewetted plots had greater cover of cottongrass *Eriophorum vaginatum* than plots that remained forested and drained. This was true after one year (clear-cut/rewetted: 5–80%; forested/drained: 1–20% cover) and after two years (clear-cut/rewetted: 20–90%; forested/drained: 1–40% cover). These results were not tested for statistical significance. Before intervention, cottongrass cover was 1% in eight of nine monitored plots (20% in the other). In February 1995, one area of a drained, tree-colonized fen was restored: trees were felled and removed, drainage ditches were filled or blocked, and an additional input ditch was excavated above the fen. In the restored area, the water table was 5–45 cm below the peat surface (during summer). The rest of the fen was left forested and drained (water table 20–65 cm below surface). Vegetation cover was visually estimated in 1994, 1995 and 1996, in six 60 x 60 cm plots in the clear-cut/rewetted area and three plots outside.

A replicated, paired, controlled study in two afforested blanket bogs in Scotland, UK (2) reported that plots restored by tree felling and rewetting had greater cover of sheathed cottongrass *Eriophorum vaginatum*, and less cover of forest mosses, than plots that remained forested and drained. These results were not tested for statistical significance. After five years, restored plots had sheathed cottongrass cover of 44–45% (vs 11% in plots that remained forested and drained), plait moss *Hypnum cupressiforme* cover of 18–32% (vs 57%) and waved silk moss *Plagiothecum undulatum* cover of <1% (vs 6%). The combined effect of felling and rewetting was larger than the effect of felling or rewetting alone in 10 of 12 comparisons. Between 1996 and 1998, six blocks of six 40 x 100 m plots were established in drained, conifer-forested bogs. Each treatment was replicated once/block: rewetting with tree felling (debris left in place), rewetting with tree felling (debris removed), rewetting only, tree felling only, tree removal only, no intervention. Rewetting was achieved by damming plough furrows every 20 m. In rewetted plots, the water table was 8–32 cm below the peat surface during the growing season (vs drained plots: 11–38 cm below). Vegetation cover was recorded five years after intervention (details not reported).

A before-and-after, site comparison study in 1994–2005 in two peatlands (one bog, one fen) in Finland (3) found that restoration by tree removal and rewetting increased the abundance of wetland-characteristic plant species and some key peatland species. The overall plant community composition changed over ten years. Restored (clear-cut/rewetted) areas accumulated wetland-characteristic species, whilst unrestored (forested/drained) areas accumulated dryland- and forest-characteristic species (data reported as a graphical analysis). Specifically, cover of sheathed cottongrass *Eriophorum vaginatum* increased more in restored areas (from 4–11% before restoration to 20–21% ten years after) than unrestored areas (from 1–3% to 1–7%). In the fen, forest moss cover decreased in the restored area (from 17 to 4%) but increased in the unrestored area (from 26 to 42%). In the bog, *Sphagnum* moss cover increased more in the restored area (data not reported). In 1995, 10 ha of bog and 1 ha of fen were restored by clearing all trees/shrubs and blocking drainage ditches. In the restored areas, the water table was 5–17 cm below the peat surface (summer average). Comparisons were made with unrestored areas in each peatland (forested and drained; water table 23–45 cm below surface). In July 1994–1997 and 2005, vegetation cover was estimated in 9–12 permanent quadrats (0.5 m^2) in each area.

A replicated before-and-after study in 2002–2005 in two drained, tree-colonized, rich fens in Sweden (4) reported that following rewetting and tree removal, there were small changes in plant community composition and cover. These results are not based on tests of statistical significance. The overall composition of the plant community changed following tree removal and rewetting (data reported as a graphical analysis). In one fen, cover of purple moor grass *Molinia caerulea* was 50% before intervention but 30% three years after. Common cottongrass *Eriophorum angustifolium* disappeared (having had 0.1% cover before intervention). Across both fens, cover of sedges *Carex* spp. was 0–1% before but 1–8% after. Cover of common reed *Phragmites australis* and *Sphagnum* mosses showed mixed responses by site or species respectively. In plots that had trees removed without rewetting, moor grass and sedge cover changed much less than under the combined treatment, whilst cottongrass cover increased (from <1 to 5%). Around winter 2002/2003, two 50 x 150 m plots (one plot/fen) were cleared of trees and rewetted by blocking drainage ditches (water table raised approximately 10 cm). Two adjacent plots (one plot/fen) were cleared of trees but remained drained. Vegetation cover was estimated before (2002) and after (2005) intervention, in 4–16 quadrats (each 0.25 m^2) in the centre of each plot. This study was based on the same experimental set-up as (6) and (8).

A replicated, before-and-after, site comparison study in 2006–2009 in 19 forested fens in Finland (5) found that restoration by tree thinning and rewetting did not affect the overall plant community, the number of plant species or cover of individual plant groups – except cottongrasses/sedges. The overall composition of the plant community was similar before and 18 months after restoration, and was similar in restored and natural fens (data reported as a graphical analysis). After 18 months, restored fens contained 44 plant species in total (vs 45 before restoration and 49 in natural fens). These results were not tested for statistical significance. Although cover of some plant groups changed significantly in restored fens (shrub cover from 33 to 42%; forb cover from 8 to 11%; cottongrass/sedge cover from 8 to 10%; *Sphagnum* moss cover from 64 to 55%; forest moss cover from 21 to 17%), the changes were mirrored in natural fens (with the exception of cottongrass/sedge cover, which decreased from 13 to 8%). In late 2007, eleven drained, densely forested fens were restored by filling drainage ditches (water table 8–16 cm below peat surface, on average, during summer) and thinning trees (from 940 to 317 stems/ha). Comparisons were made with eight nearby natural fens (water table 8–17 cm below surface; 373 trees/ha). In July 2006 and 2009, cover of every plant species was estimated in approximately twelve 1 m^2 quadrats/fen.

A replicated, paired, controlled, before-and-after study in 2002–2010 in three drained, tree-colonized, rich fens in Sweden (6) reported that following tree removal and rewetting, there were increases in plant species richness, bryophyte cover and sedge cover. These results are not based on tests of statistical significance. There were 9 plant species/0.25 m^2 before intervention but 13 species/0.25 m^2 eight years after. Cover of wetland-characteristic bryophytes was 33% before and 46% after, *Sphagnum* mosses 23% before and 33% after, sedges 1% before and 5% after. Similar changes in cover occurred in plots that were rewetted (without tree removal) or had trees removed (without rewetting). In control plots that remained both drained and forested, there was no change in the number of plant species or vegetation cover. In winter 2002/2003, four restoration treatments were applied in each drained and tree-colonized fen, in adjacent 50 x 150 m plots: cutting and removal of all trees, rewetting (by ditch blocking; water table raised by 12–25 cm), tree removal and rewetting, or none. Between 2002 (before intervention) and 2010, cover of every plant species was estimated at 20 points/plot, in 0.25 m^2 quadrats. This study was based on the same experimental set-up as (4) and (8).

A replicated site comparison study in 2011 across 21 blanket bogs in Scotland, UK (7) found that restoration by tree felling and rewetting increased vegetation height and cover of grass-like herbs. After 5–13 years, restored bogs had significantly taller ground vegetation (21 cm) than forested/drained bogs (3 cm) and naturally open bogs (17 cm). Amongst restored sites, vegetation height declined with time since restoration (see original paper for data and statistical model). Restored bogs had significantly greater cover of grass-like herbs than forested/drained sites (mostly bare ground covered in pine needles) and natural bogs (moss-dominated; data and species not reported). In summer 2011, twenty-one bogs were surveyed: eight restored (conifers cut and drainage ditches blocked 5–13 years before surveying, raising the water table "close to the ground surface"), six degraded (conifer-forested/drained) and seven natural (unforested/undrained). In each bog, ground vegetation height (i.e. excluding trees) was measured at 45 points, distributed along fifteen 10 m transects. Details of cover measurements were not reported.

A replicated, paired, controlled, site comparison study in 2002–2010 involving three degraded rich fens in Sweden (8) reported that clear-cut and rewetted plots developed greater plant species richness than plots that remained forested and drained, but that vegetation grew to a similar height. Most of these results were not tested for statistical significance. After eight years, clear-cut and rewetted plots contained 14 species/0.25 m^2, compared to 9 species/0.25 m^2 in plots that remained forested and drained. Canopy height (of vegetation that grew following intervention) in clear-cut/rewetted plots was 5 m, compared to 6 m in drained/forested plots. Plots only rewetted *or* cleared had similar species richness (13–14 species/0.25 m^2) and vegetation height (5–6 m) to the plots both rewetted *and* cleared. For comparison, a nearby natural (undrained and unforested) fen contained 9 plant species/0.25 m^2 and had a canopy height of 1 m. These were significantly greater in the clear-cut/rewetted plots. Around winter 2002/2003, four restoration treatments were applied in each drained and tree-colonized fen, in adjacent 50 x 150 m plots: cutting and removal of all trees, rewetting (by ditch blocking), tree removal and rewetting, or none. In 2010, plant species and canopy height were recorded at 20 points/plot, in 0.25 m^2 quadrats. The natural fen was sampled in 1978. This study was based on the same experimental set-up as (4) and (6).

A replicated, paired, site comparison study in 2003–2007 in nine bogs and fens in Finland (9) reported that areas restored by tree thinning and rewetting had moss cover and tree structure intermediate between degraded (forested and drained) and natural (sparse trees, never drained) areas. After 1–3 years, restored areas had greater *Sphagnum* moss cover but less cover of other mosses than degraded areas, but less *Sphagnum* moss cover and greater cover of other mosses than natural areas. Restored areas had fewer tall trees (>3m) than degraded areas, but more tall trees than natural areas. All data were reported as graphical analyses and differences were not tested for statistical significance. Between 2003 and 2006, in each of nine degraded peatlands, one area was managed by removing excess trees (above the natural tree density) and blocking drainage ditches. In each peatland, one degraded and one pristine area were also monitored. In 2007, vegetation cover was visually estimated in twenty-four 1 m^2 quadrats/area (72 quadrats/peatland). Trees were counted and measured in six 100 m^2 plots/area (18 plots/peatland). This study was based on the same experimental set-up as (11).

A replicated site comparison study in 2007 in 19 forested fens in Finland (10) found that restoration by tree thinning and rewetting had no effect on plant taxon richness, diversity or community composition. After 3–12 years, there were no significant differences between treatments for plant taxon richness (restored: 11–12; degraded: 11; natural: 12 taxa/m^2) or diversity (data reported as a diversity index). Overall plant community composition did not differ between restored and degraded sites, but was significantly different from natural sites in both (data reported as a graphical analysis). Of the 19 studied forested fens, 10 had been restored 3–12 years before sampling (trees thinned and drainage ditches filled; water table 16 cm below peat surface). Four fens were degraded (excess tree growth and drained; water table 32 cm below surface). Five fens were natural (sparsely forested and undrained; water table 19 cm below surface). In summer 2007, cover of plant taxa was estimated in three 1 m^2 plots at each site.

A replicated, paired, before-and-after, site comparison study in 2003–2007 in nine bogs and fens in Finland (11) reported that a combination of tree thinning and rewetting reduced the number of tall trees for 1–3 years. Areas that were rewetted and cleared of trees contained fewer tall (>3 m) trees 1–3 years after restoration than before. Thus, the number of tall trees in restored areas became more like natural areas and less like degraded areas. Data were reported as graphical analyses and differences were not tested for statistical significance. Between 2003 and 2006, in each of nine peatlands, one area previously drained for forestry was restored by removing excess trees (above the natural tree density) and blocking drainage ditches. This was compared to one area that remained degraded (drained and fully forested) and one pristine area (never drained, sparsely forested). In 2003 (before intervention) and 2007, trees were counted and measured in six 100 m^2 plots/area (18 plots/peatland). This study was based on the same experimental set-up as (9).

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(2) Anderson R. (2010) *Restoring afforested peat bogs: results of current research*. Forestry Commission Research Note 6.
(3) Haapalehto T.O., Vasander H., Jauhiainen S., Tahvanainen T. & Kotiaho J.S. (2010) The effects of peatland restoration on water-table depth, elemental concentrations, and vegetation: 10 years of changes. *Restoration Ecology*, 19, 587–598.
(4) Mälson K., Sundberg S. & Rydin H. (2010) Peat disturbance, mowing, and ditch blocking as tools in rich fen restoration. *Restoration Ecology*, 18, 469–478.
(5) Laine A.M., Leppälä M., Tarvainen O., Päätalo M.L. Seppänen, R. & Tolvanen A. (2011) Restoration of managed pine fens: effect on hydrology and vegetation. *Applied Vegetation Science*, 14, 340–349.
(6) Hedberg P., Kotowski W., Saetre P., Mälson K., Rydin H. & Sundberg S. (2012) Vegetation recovery after multiple-site experimental fen restorations. *Biological Conservation*, 147, 60–67.
(7) Gilbert L. (2013) Can restoration of afforested peatland regulate pests and disease? *Journal of Applied Ecology*, 50, 1226–1233.
(8) Hedberg P., Saetre P., Sundberg S., Rydin H. & Kotowski W. (2013) A functional trait approach to fen restoration analysis. *Applied Vegetation Science*, 16, 658–666.

(9) Noreika N., Kotiaho J.S., Penttinen J., Punttila P., Vuori A., Pajunen T., Autio O., Loukola O.J. & Kotze D.J. (2015) Rapid recovery of invertebrate communities after ecological restoration of boreal mires. *Restoration Ecology*, 23, 566–579.
(10) Daza Secco E., Haapalehto T., Haimi J., Meissner K. & Tahvanainen T. (2016) Do testate amoebae communities recover in concordance with vegetation after restoration of drained peatlands? *Mires and Peat*, 18, Article 12.
(11) Noreika N., Kotze D.J., Loukola O.J., Sormunen N., Vuori A., Päivinen J., Penttinen J., Punttila P. & Kotiaho J.S. (2016) Specialist butterflies benefit most from the ecological restoration of mires. *Biological Conservation*, 196, 103–114.

N.B. Results from (1) are also reported in: Komulainen V.-M., Nykänen H., Martikainen P.J. & Laine J. (1998) Short-term effect of restoration on vegetation change and methane emissions from peatlands drained for forestry in southern Finland. *Canadian Journal of Forest Research*, 28, 402–411.

Interventions: Livestock farming and ranching

3.5 Use barriers to keep livestock off ungrazed peatlands Ⓑ Ⓕ S

- We captured no evidence for the effect on peatland vegetation of using barriers to keep livestock off peatlands that have never (or not recently) been grazed.

Background

This section considers the effects of excluding livestock (with fences or other barriers) from an area of natural, ungrazed peatland whilst the surrounding peatland becomes grazed. This means the exclusion area has never (or at least not recently) been grazed. Domestic livestock directly consume peatland vegetation, destroy peatland vegetation by trampling, create bare patches of ground (e.g. repeatedly used tracks), and affect nutrient balance through excretion (Lindsay *et al.* 2014).

Related interventions: removing or excluding livestock from *degraded* peatlands (Sections 3.6 and 3.7); rewetting, if peatland has been drained for agriculture (Section 8.1); low intensity grazing as a conservation tool (Sections 8.8 and 9.7); exclude wild herbivores from peatlands (Section 9.12); use fences or barriers specifically to protect planted/sown peatland plants (Section 13.15).

Lindsay R., Birnie R. & Clough J. (2014) *Grazing and Trampling*. IUCN UK Peatland Programme Briefing Note No. 7.

3.6 Exclude or remove livestock from degraded peatlands Ⓑ Ⓕ S

- **Ten studies** examined the effect on peatland vegetation of excluding or removing livestock from degraded peatlands. Seven studies were in bogs[1a,1b,2,3,5,6,7], two in fens[4,9] and one in an unspecified peatland[8]. Three studies[1b,6,7] were based on the same experimental set-up in the UK.
- **Plant community composition (2 studies):** Of two replicated, paired, controlled studies in bogs in the UK, one[7] found that excluding sheep had no effect on the development of the plant community. The other[5] found no effect in wetter areas of the bog, but that in drier areas excluding sheep favoured dry moorland plants.
- **Herb cover (9 studies):** Seven studies (including six replicated, paired, controlled) in bogs in the UK[1a,1b,2,5,7] and Australia[3] and fens in the USA[4] found that excluding or removing livestock had no

effect on cover of key herb groups. Five of five studies[1a,1b,2,5,7] found that excluding livestock typically had no effect on cottongrass cover. Two of two[2,4] studies reported no effect on sedge cover. However, one before-and-after study in a poor fen in Spain[9] reported that rush cover increased after cattle were excluded (along with other interventions). One site comparison study in Chile[8] found that excluding livestock (along with other interventions) increased overall herb cover, but one replicated, paired, controlled study in bogs in Australia[3] found that excluding livestock had no effect on overall herb cover.

- **Moss cover (6 studies):** Five replicated, paired, controlled studies in bogs in the UK[1a,1b,2,5] and Australia[3] found that excluding livestock typically had no effect on *Sphagnum* moss cover. Responses sometimes varied between species and sites. Three of the studies in the UK[1a,1b,2] also found no effect on cover of other mosses. One before-and-after study in a poor fen in Spain[9] reported that *Sphagnum* moss appeared after excluding cattle (and rewetting).
- **Tree/shrub cover (8 studies):** Four replicated, paired, controlled studies in bogs in the UK[1a,1b,7] and Australia[3] found that excluding livestock had no effect on shrub cover (specifically heather[1a,1b,7] or a heathland community[3]). One replicated, paired, controlled study a bog in the UK[5] found that excluding sheep had no effect on heather cover in wetter areas, but increased heather cover in drier areas. Three studies (including two site comparisons) in bogs in the UK[2], fens in the USA[4] and a peatland in Chile[8] found that excluding or removing livestock increased shrub cover.
- **Vegetation structure (1 study):** One replicated, paired, controlled study in a bog in the UK[6] found that excluding sheep increased total vegetation, shrub and bryophyte biomass but had no effect on biomass of grass-like herbs.

Background

This section considers management of peatland vegetation by completely excluding or removing livestock from degraded peatlands.

Domestic livestock directly consume peatland vegetation, destroy peatland vegetation by trampling, create bare patches of ground (e.g. repeatedly used tracks), and affect nutrient balance through excretion (Lindsay *et al.* 2014). Some plant groups or species, such as heather and dwarf shrubs, may be impacted more by selective grazing (Grant *et al.* 1987). Removing livestock could allow these species to recover to natural levels, although there is a risk that they become over-abundant when not grazed.

Related interventions: excluding livestock from *pristine or undisturbed* peatlands (Section 3.5); reducing grazing intensity, rather than completely removing livestock (Section 3.7); rewetting if peatland has been drained for agriculture (Section 8.1); low intensity grazing as a conservation tool (Sections 8.8 and 9.7); excluding wild herbivores from peatlands (Section 9.12); using fences or barriers specifically to protect planted/sown peatland plants (Section 13.15).

Grant S.A., Suckling S.A., Smith H.K., Torvell L., Forbes T.D.A. & Hodgson J. (1987). Comparative studies of diet selection by sheep and cattle: blanket bog and heather moor. *Journal of Ecology*, 75, 947–960.

Lindsay R., Birnie R. & Clough J. (2014) *Grazing and Trampling*. IUCN UK Peatland Programme Briefing Note No. 7.

A replicated, paired, controlled, before-and-after study in 1953–1975 in a grazed blanket bog in England, UK (1a) found that excluding sheep typically had no effect on heather *Calluna vulgaris* or cottongrass *Eriophorum* spp. cover, but had mixed effects on moss cover. In two of three sites, plots from which sheep had been excluded had similar cover to grazed plots of heather (exclusion: 60–70%; grazed: 50–60%) and cottongrasses (exclusion: 7%; grazed: 7%). Meanwhile in a third site, around a bog pool, cover increased over 21 years of sheep exclusion for both heather (from 40 to 54%) and cottongrasses (from 0 to 2%). Moss cover showed mixed responses to sheep exclusion: both *Sphagnum* and other moss cover were lower in exclusion plots than grazed plots in one site (exclusion: 3%; grazed: 4–5%), similar in exclusion and grazed plots in one site (exclusion: 1%; grazed: 1%), but showed mixed responses by species around the bog pool. In 1953 or 1968, sheep were excluded from part of each site with 20 cm mesh fencing. The rest of each site remained grazed (<0.3 sheep/ha). Vegetation cover was recorded after 7, 18 or 12 years of exclosure (and immediately before exclosure in the bog pool site).

A replicated, randomized, paired, controlled study in 1954–1973 in a grazed and recently burned blanket bog in England, UK (1b) found that excluding sheep typically had no effect on vegetation cover, but did increase the number of heather *Calluna vulgaris* shoots and stems. For 32 of 37 plant groups, cover never significantly differed between plots from which sheep had been excluded and plots that remained grazed. These included *Sphagnum* mosses (exclusion: 5–19%; grazed: 2–8%), six of seven other moss species (exclusion: 1–38%; grazed: 1–46%), cottongrasses *Eriophorum* spp. (exclusion: 6–62%; grazed: 9–67%) and live heather (exclusion: 30–82%; grazed: 19–70%). However, exclusion plots did contain a greater density of heather shoots and stems than grazed plots. In 1954, four areas of a grazed bog (<0.3 sheep/ha) were burned once. Within each area, a random three of six 1,000 m^2 plots were fenced to exclude sheep. In each area, one fenced and one unfenced plot were burned again in 1965. In 1972, vegetation cover was estimated by recording, in each plot, plants touching 250 randomly placed pins. This study was based on the same experimental set-up as (6) and (7).

A replicated, paired, controlled, before-and-after study in 1966–1980 in two grazed blanket bogs in England, UK (2) found that excluding sheep increased shrub cover, but typically had no effect on cover of moss or herb species. In exclusion plots, cover increased of heather *Calluna vulgaris* (before: 0–4%; after 14 years: 2–21%) and crowberry *Empetrum nigrum* (before: 7–17%; after: 27–42%). These changes were significant in two of four comparisons, with a similar trend in the others. Cover of two other shrub species did not decline. There was no significant change in cover of six of six moss species, including two *Sphagnum* (before: 0–10%; after: 1–21%) or in 8 of 11 comparisons involving herb species, including black sedge *Carex nigra* (before: 6%; after: 4%) and cottongrasses *Eriophorum* spp. (before: 18–72%; after: 16–69%). Vegetation cover generally did not change in grazed control plots (except for a decrease in heather cover in one site, from 6% to 3%). In 1966, 0.1 ha of each bog was fenced to exclude sheep. An adjacent plot in each bog, with similar vegetation, was left open to grazing (<0.5 sheep/ha). In 1966 and 1980, vegetation cover was measured using 500 systematically placed pins in each plot.

A replicated, paired, controlled study in 1980–1996 in two grazed bogs in Australia (3) found that excluding cattle with fences had no effect on vegetation cover. Over 15 years, cover of different vegetation types changed similarly in exclusion and grazed plots. Although *Sphagnum* moss cover increased in exclusion plots (from 15–20% to 23–24%), it also increased in grazed plots (from 16–18% to 19–20%). In one bog where heathland vegetation cover increased, it did so in both exclusion plots (from 5 to 30%) and grazed plots (from 4 to 23%). Herb cover did not change in exclusion plots (from 54–77% to 54–70%) or grazed plots (from 49–81% to 52–77%). In 1980–1981, one pair of plots was established in each grazed bog. Two plots (one plot/pair) were fenced to exclude free-ranging cattle. The other plots remained open to grazing. In 1981 and 1996, vegetation cover was recorded along 5–15 transects/bog, each 20–70 m long.

A site comparison study in 1977–1997 in three historically grazed sedge meadows in Wisconsin, USA (4) reported that after cattle grazing was stopped, vegetation structure and species richness became more like a meadow that had never been grazed, but shrub cover less so. Most of these results were not tested for statistical significance. Between four and twenty years after grazing was stopped, average vegetation cover and height increased (data reported as graphical analyses) whilst the number of plant species decreased (from 43 to 34). For these measures, the previously grazed meadow became more like a never-grazed meadow and less like a heavily grazed meadow. In contrast, cover of red twig dogwood *Cornus sericea* increased significantly in the previously grazed meadow (from 0 to 9%) but not in the never-grazed meadow (from 0 to 2%). Total sedge *Carex* spp. cover did not change significantly over time in any site. In 1977 and 1997, three sedge meadows were studied: one previously grazed (heavily grazed until 1973, when grazing stopped), one that remained heavily grazed, and one effectively never grazed (lightly grazed until the 1960s). Sedge meadows are sedge-dominated peatlands, fed by ground water. Cover and height of every plant species were recorded in 20–28 quadrats (0.2 m^2) per meadow.

A replicated, paired, controlled study in 1988–2002 in a grazed bog in England, UK (5) found that excluding sheep changed the plant community composition and vegetation cover in drier parts of the bog, but had no effect in wetter parts of the bog. Exclusion and grazed plots developed different plant communities over 14 years in drier areas, but retained similar communities to each other in wetter areas (data reported as graphical analyses). After 14 years, exclusion plots in dry areas had greater cover of heather *Calluna vulgaris* than grazed plots (exclusion: 7%; grazed: 1%) and less cover of Magellan's bog moss *Sphagnum magellanicum* (exclusion: 8%; grazed: 23%). In both wet and dry areas, excluding sheep did not affect cover of other common plant species including cottongrasses *Eriophorum* spp. (exclusion: 4–23%; grazed: 6–19%) and other *Sphagnum* moss species (exclusion: 4–21%; grazed: 3–36%). In 1988, ten pairs of 20 x 20 m^2 plots were established in a grazed bog: five pairs in the wetter central part of the bog and five pairs in the drier margins. Five plots (one plot/pair) were fenced to exclude sheep. The other plots remained grazed (0.65 sheep/ha). In 1988 and 2002, vegetation cover was visually estimated in ten 1 m^2 quadrats/plot.

A replicated, paired, controlled study in 1954–2004 in a grazed and recently burned blanket bog in England, UK (6) found plots fenced to exclude sheep contained more total vegetation, shrub and bryophyte biomass than grazed plots, but similar biomass of grass-like herbs. After 50 years, above-ground vegetation biomass was greater in exclusion plots (240 g/m^2) than grazed plots (192 g/m^2). This included greater biomass of shrubs (mainly heather *Calluna vulgaris*; exclusion: 194; grazed: 161 g/m^2) and bryophytes (mainly red-stemmed feather moss *Pleurozium schreberi*; exclusion: 37; grazed: 18 g/m^2). However, exclusion and grazed plots contained similar biomass of grass-like herbs (mainly sheathed cottongrass *Eriophorum vaginatum*; exclusion: 10; grazed: 13 g/m^2). In 1954, sixteen 1,000 m^2 plots were established (in four blocks of four) on a grazed bog. Eight plots (two plots/block) were fenced to exclude sheep. The other eight plots remained open to summer grazing (0.04 sheep/ha). All plots were burned once in 1954, with half also burned every 10 years thereafter. In 2003–2004, live above-ground vegetation was cut from one 25 cm^2 quadrat/plot, then dried and weighed. Samples were taken in spring, summer, autumn and winter. This study was based on the same experimental set-up as (1b) and (7).

A replicated, randomized, paired, controlled study in 1954–2001 in a grazed and recently burned blanket bog in England, UK (7) found that excluding sheep had no effect on plant community composition, cover and species richness. Between 1972 and 2001, the overall plant community composition changed in both grazed and ungrazed areas, from liverwort-rich to heather- or cottongrass-rich (depending on whether they were also burned). However, community development was not significantly affected by grazing (data reported as graphical analyses). Over the experimental period, exclusion and grazed plots contained a similar number of plant species and *Sphagnum* moss species, and similar cover of heather *Calluna vulgaris* and cottongrasses *Eriophorum* spp. (amongst other species; data not reported). In 1954, four 60 x 90 m areas of a grazed bog received an initial burn. Then, sheep were excluded from half of each area whilst the other half remained open to grazing (0.1–0.3 sheep/ha). Of three plots within each grazed and ungrazed area, two were burned again during the study period (every 10 or 20 years). Vegetation cover was measured in 1972, 1982, 1991 and 2001 by recording, in each area, plants touching 300 randomly placed pins. This study was based on the same experimental set-up as (1b) and (6).

A site comparison study in 2014 in a peatland in Chile (8) found that a protected area (fenced to exclude livestock and where moss harvesting was prohibited) had greater vegetation cover and taller vegetation, but lower vascular plant richness and diversity, than an adjacent unprotected (grazed and harvested) area. The protected area had greater cover than the unprotected area of total vegetation (87 vs 62%), herbs (68 vs 51%) and shrubs (19 vs 11%) and contained taller vegetation (65 vs 13 cm). The protected area had lower vascular plant species richness than the unprotected area (7 vs 11 species/4 m^2) and lower diversity (reported as a diversity index), but also contained fewer non-native species (<0.1 vs 1.9 species/4 m^2). In 2014, vegetation cover and height were recorded in forty-four 2 x 2 m quadrats. Fifteen quadrats were in 5.5 ha of protected peatland, fenced to exclude oxen for eight years and with no moss harvesting for at least 20 years. The study does not distinguish between the effects of these interventions. Twenty-nine quadrats were in 10.5 ha of unprotected peatland, grazed by four oxen and harvested monthly.

A before-and-after study in 2008–2013 in a historically grazed poor fen in Spain (9) reported that after building fences to exclude cattle (along with rewetting), cover of rushes *Juncus* spp. increased and new populations of *Sphagnum* moss appeared. No statistical tests were carried out. Before intervention, the fen was covered by dryland grasses and forbs, with no *Sphagnum*. Four years after intervention, 81% of the peatland area contained rushes: common rush *Juncus effusus* with some sharp-flowered rush *Juncus acutiflorus*. *Sphagnum* mosses also appeared in 3 of 10 monitored quadrats. In 2009, fences were built to exclude cattle. At the same time, the fen was rewetted by blocking/removing drainage channels and building a new inflow ditch. The study does not distinguish between the effects of cattle exclusion and rewetting. Vegetation cover before (2008) and after (2013) intervention was mapped from aerial photos and recorded in ten permanent quadrats (size not reported).

(1) Rawes M. & Hobbs R. (1979) Management of semi-natural blanket bog in the northern Pennines. *Journal of Ecology*, 67, 789–807.
(2) Rawes M. (1983) Changes in two high altitude blanket bogs after the cessation of sheep grazing. *Journal of Ecology*, 71, 219–235.
(3) Wahren C.-H.A., Williams R.J. & Papst W.A. (2001) Vegetation change and ecological processes in alpine and subalpine *Sphagnum* bogs of the Bogong High Plains, Victoria, Australia. *Arctic, Antarctic and Alpine Research*, 33, 357–368.
(4) Middleton B. (2002) Nonequilibrium dynamics of sedge meadows grazed by cattle in southern Wisconsin. *Plant Ecology*, 161, 89–110.
(5) Smith R.S., Charman D., Rushton S.P., Sanderson R.A., Simkin J.M. & Shiel R.S. (2003) Vegetation change in an ombrotrophic mire in northern England after excluding sheep. *Applied Vegetation Science*, 6, 261–270.
(6) Ward S.E., Bardgett R.D., McNamara N.P., Adamson J.K. & Ostle N.J. (2007) Long-term consequences of grazing and burning on northern peatland carbon dynamics. *Ecosystems*, 10, 1069–1083.
(7) Lee H., Alday J.G., Rose R.J., O'Reilly J. & Marrs R.H. (2013) Long-term effects of rotational prescribed burning and low-intensity sheep grazing on blanket-bog plant communities. *Journal of Applied Ecology*, 50, 625–635.
(8) Cabezas J., Galleguillos M., Valdés A., Fuentes J.P., Pérez C. & Perez-Quezada J.F. (2015) Evaluation of impacts of management in an anthropogenic peatland using field and remote sensing data. *Ecosphere*, 6, 1–24.
(9) Peralta de Andrés J., Heras Pérez P., Infante Sánchez M. & Berastegi Gartziandia A. (2015) Cambios de la vegetación tras la restauración de la turbera de Belate (Navarra) observados mediante cartografía diacrónica: 2008–2013 (Vegetation changes after restoration of Belate peatland observed by diachronic mapping; in Spanish). Pages 1823–1831 in: J. de la Riva, P. Ibarra, R. Montorio & M. Rodrigues (eds.) *Análisis espacial y representación geográfica: innovación et aplicación*. Universidad de Zaragoza, Spain.

3.7 Reduce intensity of livestock grazing

Ⓑ Ⓕ S

- **One study** examined the effect on peatland vegetation of reducing livestock grazing intensity. This study was in bogs.
- **Vegetation cover (1 study):** One replicated, paired, controlled study in bogs in the UK[1] found that total vegetation and shrub cover were greater where grazing intensity was lower. Cottongrass cover was greater where grazing intensity was lower (one species) or unaffected by grazing intensity (one species).
- **Vegetation structure (1 study):** The same study[1] found that vegetation biomass was higher where grazing intensity was lower.

Background

This section considers the effects of reducing the intensity of grazing, but not completely removing livestock from peatlands. Grazing intensity could be reduced by letting fewer animals graze, allowing them to graze for fewer days (rotational or seasonal grazing), providing supplementary food as an alternative to living plants and/or encouraging use of alternative sites (e.g. through placement of feeding stations or shelter).

Domestic livestock directly consume peatland vegetation, destroy peatland vegetation by trampling, create bare patches of ground (e.g. repeatedly used tracks), and affect nutrient balance through excretion (Lindsay *et al.* 2014). Lower intensity grazing may have less of an impact and may maintain a more desirable plant community (Middleton *et al.* 2006). Plant groups or species that are most affected by trampling or are selectively grazed (Grant *et al.* 1987) could to recover to more desirable levels.

Maintaining some grazing may prevent any one species from becoming over-abundant.

Related interventions: completely remove livestock from degraded peatlands (Section 3.6); low intensity grazing as a conservation tool (Sections 8.8 and 9.7); restoration of damaged peatlands (Chapter 12).

Grant S.A., Suckling S.A., Smith H.K., Torvell L., Forbes T.D.A. & Hodgson J. (1987). Comparative studies of diet selection by sheep and cattle: blanket bog and heather moor. *Journal of Ecology*, 75, 947–960.

Lindsay R., Birnie R. & Clough J. (2014) *Grazing and Trampling*. IUCN UK Peatland Programme Briefing Note No. 7.

Middleton B., Holsten B. & van Diggelen R. (2006) Biodiversity management of fens and fen meadows by grazing, cutting and burning. *Applied Vegetation Science*, 9, 307–316.

A replicated, paired, controlled study in 1971–1982 in three recently burned blanket bogs in Scotland, UK (1) found that plots under lower grazing intensities had greater vegetation biomass and cover than more heavily grazed plots. After six years, vascular plant above-ground biomass was greater in lightly/moderately grazed plots (550–557 g/m^2) than in heavily grazed plots (346 g/m^2). After 11 years, the lightly/moderately grazed plots also had greater total vegetation cover (light: 81%; moderate: 69%; heavy: 48%), shrub cover (light: 13–53%; moderate: 9–38%; heavy: 8–26%) and sheathed cottongrass *Eriophorum vaginatum* cover (light: 15%; moderate: 11%; heavy: 6%). Cover of common cottongrass *Eriophorum angustifolium* was similar under all grazing intensities (data not reported). From August 1971, one 0.1 ha plot/bog was grazed under each intensity: light (136–237 sheep grazing days/ha/yr), moderate (296–494) or heavy (484–810). Between 1971 and 1980, dry above-ground biomass was measured in ten quadrats (approximately 25 x 50 cm) per plot. In 1972 and 1982, vegetation cover was measured in 20 point quadrats/plot.

(1) Grant S.A., Bolton G.R. & Torvell L. (1985) The responses of blanket bog vegetation to controlled grazing by hill sheep. *Journal of Applied Ecology*, 22, 739–751.

3.8 Change type of livestock

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- We captured no evidence for the effect on peatland vegetation of changing livestock type.

Background

Changing the species or breed of livestock on peatlands could reduce undesirable impacts. For example, using light sheep rather than heavy cows may reduce trampling impacts. Additionally, different species and breeds of livestock feed in different ways, leading to different impacts on vegetation (Loucougaray *et al.* 2004). Sheep nibble buds and shoots of selected plants, whilst cattle are less picky and pull off clumps of mixed vegetation. Sheep maintain shorter, more uniform lawns of vegetation than cattle which leave tufts of longer vegetation, whilst horses can maintain patches of short vegetation. Traditional or heritage livestock breeds may consume different species of plants in different amounts to modern breeds (Tolhurst & Oates 2001) and be better adapted to harsh conditions on some peatlands e.g. upland bogs.

Related interventions: completely remove livestock from degraded peatlands (Section 3.6); low intensity grazing for conservation (Sections 8.8 and 9.7).

Loucougaray G., Bonis A. & Bouzillé J.-B. (2014) Effects of grazing by horses and/or cattle on the diversity of coastal grasslands in western France. *Biological Conservation*, 116, 59–71.

Tolhurst S. & Oates M. (2001) The Breed Profiles' Handbook: A Guide to the Selection of Livestock Breeds for Grazing Wildlife Sites. English Nature, Peterborough.

3.9 Change season/timing of livestock grazing

Ⓑ Ⓕ S

- We captured no evidence for the effect on peatland vegetation of changing the season or timing of livestock grazing.

Background

Grazing could have different effects on peatland vegetation depending on the time of year at which it is done. For example, it might be beneficial to avoid grazing when certain plants are young/flowering so that they can grow/reproduce and contribute to the peatland vegetation. Additionally, the effects of trampling may vary by season, being lowest in summer when the peatland is relatively dry or winter if it is frozen. Clearly, seasonal variation in the value of the peatland plants as food for livestock could also contribute to the decision of when to allow grazing.

Related interventions: completely remove livestock from degraded peatlands (Section 3.6); low intensity grazing for conservation (Sections 8.8 and 9.7).

4. Threat: Energy production and mining

Background

This chapter addresses threats from the production of energy or non-living resources (especially peat). Interventions to tackle threats from energy infrastructure, such as wind turbines and roads, are considered elsewhere (Chapters 2, 5 and 12).

Peat is extracted on both industrial and subsistence scales for uses including horticulture, fuel and therapeutic treatments (Salvador *et al.* 2014). Peat extraction leaves bare exposed peat, on which vegetation recovery can be slow (Lavoie *et al.* 2003). However, peat harvesting affects only a small proportion of all peatlands: <1% of the peatland area Sweden and Finland and <4% in Estonia has been lost to peat harvesting (Vasander *et al.* 2003). Peatlands may also be excavated to access underlying minerals, metals or fossil fuels e.g. oil sands in Canada (Rooney *et al.* 2012) and diamonds in Lesotho (Grundling *et al*. 2015). For simplicity and brevity, we will use the term *mining* to describe extraction of any non-living resource from peatlands (although it typically refers to peat extraction, harvesting or excavation).

Mined peatlands could be left to recover without any active intervention. Passive recovery is cheaper than active intervention, but may be ineffective or slow. We do not assess the evidence for this in detail, but note that many mined peatlands remain bare for decades after abandonment because they are exposed to strong winds, covered by a hard dry crust and/or situated far from any source of colonizing plants (e.g. Lavoie *et al.* 2003; Konvalinkova & Prach 2010). Any vegetation that does appear might be dominated by non-native species (Domínguez *et al.* 2012).

Related threats: development, including construction of energy infrastructure (Chapter 2 and Chapter 5); biological resource use i.e. harvesting live peatland plants (Chapter 6); human intrusions and disturbance, including from vehicles (Chapter 7); natural system modifications, including flooding from hydroelectric dams (Chapter 8); pollution from mining activities (Chapter 10). Related interventions: habitat creation and restoration (Chapter 12); legal protection and lobbying against all forms of land use change, including energy production and mining (Chapter 14 and Chapter 15);

Domínguez E., Bahamonde N. & Muñoz-Escobar C. (2012) Efectos de la extracción de turba sobre la composición y estructura de una turbera de *Sphagnum* explotada y abandonada hace 20 años, Chile (Effects of peat extraction on the composition and structure of a *Sphagnum* peat bog exploited and abandoned 20 years ago; in Spanish). *Anales Instituto Patagonia* (Chile), 40, 37–45.

Grundling P.-L., Linström A., Fokkema W. & Grootjans A.P. (2015) Mires in the Maluti Mountains of Lesotho. *Mires and Peat*, 15, Article 9.

Konvalinkova P. & Prach K. (2010) Spontaneous succession of vegetation in mined peatlands: a multi-site study. *Preslia*, 82, 423–435.

Lavoie C., Grosvernier P., Girard M. & Marcoux K. (2003) Spontaneous revegetation of mined peatlands: an useful restoration tool? *Wetlands Ecology and Management*, 11, 97–107.

Rooney R.C., Bayley S.E. & Schindler D.W. (2012) Oil sands mining and reclamation cause massive loss o peatland and stored carbon. *Proceedings of the National Academy of Sciences USA*, 109, 4933–4937.

Salvador F., Monerris J. & Rochefort L. (2014) Peatlands of the Peruvian Puna ecoregion: types, characteristics and disturbance. *Mires and Peat*, 15, Article 3.

Vasander H., Tuittila E.-S., Lode E., Lundin L., Ilomets M., Sallantaus T., Heikkilä R., Pitkänen H.-L. & Laine J. (2003) Status and restoration of peatlands in northern Europe. *Wetlands Ecology and Management*, 11, 51–63.

Key messages

4.1 *Replace blocks of vegetation after mining or peat extraction* **2 studies**

Plant community composition: Two studies, in bogs in the UK and a fen in Canada, reported that transplanted blocks of peatland vegetation retained their overall community composition: over time in the UK, or relative to an undisturbed fen in Canada.

Vegetation cover: One before-and-after study in the UK reported that bare peat next to translocated bog vegetation developed vegetation cover (mainly grasses/rushes). *Sphagnum* moss cover declined in the translocated blocks. One site comparison study in a fen in Canada reported that replaced vegetation blocks retained similar *Sphagnum* and shrub cover to an undisturbed fen.

4.2 *Retain/create habitat corridors in areas of energy production or mining* **0 studies**

We captured no evidence for the effect on peatland vegetation, in habitat patches or within corridors, of retaining/creating habitat corridors in areas of energy production or mining.

Interventions

4.1 Replace blocks of vegetation after mining or peat extraction

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- **Two studies** examined the effect on peatland vegetation of replacing blocks of vegetation after mining or peat extraction. One study was in a bog[1] and one was in a fen[2].
- **Plant community composition (2 studies):** Two studies, in a bog in the UK[1] and a fen in Canada[2], reported that transplanted vegetation blocks retained their peatland vegetation community. In the UK[1], the community of the transplanted blocks did not change over time. In Canada[2], the community of replaced vegetation blocks remained similar to an undisturbed fen.
- **Vegetation cover (2 studies):** One before-and-after study in the UK[1] reported that bare peat next to transplanted bog vegetation developed vegetation cover (mainly grass/rush). *Sphagnum* moss cover declined in the translocated blocks. One site comparison study in Canada[2] reported that replaced fen vegetation retained similar *Sphagnum* and shrub cover to an undisturbed fen.

Background

This intervention involves replacing blocks of peatland vegetation on a bare peat surface, such as that left behind after mining subsurface peat. In this way, the surface vegetation may recover much more quickly than if reassembled through colonization or planting individual plants. Blocks of introduced vegetation with an intact surface layer may help to regulate moisture.

The vegetation blocks should be cut from the upper 30 cm of natural peatlands. They may be cut *from the mined peatland* before mining begins, then kept aside during mining. This retention and replacement approach has its origins in the German *Bunkerde* concept (Money & Wheeler 1999) but is also known as 'peat-block reclamation' (Cagampan & Waddington 2008). Alternatively, vegetation blocks may be *cut from another peatland area* and transplanted to the damaged area, as in 'bofedal transplants' used in South America. Clearly, this damages to the donor bog (although blocks could be sourced from peatlands destined to be destroyed by development).

Related intervention: rewetting, because peatlands are often drained to allow peat extraction (Sections 8.1 and 14.7).

Cagampan J.P. & Waddington J.M. (2008) Moisture dynamics and hydrophysical properties of a transplanted acrotelm on a cutover peatland. *Hydrological Processes*, 22, 1776–1787.

Money R.P. & Wheeler B.D. (1999) Some critical questions concerning the restorability of raised bogs. *Applied Vegetation Science*, 2, 107–116.

A before-and-after study in 1991–1997 in a historically mined blanket bog/heathland in England, UK (1) reported that translocated bog vegetation retained its overall community composition whilst gaining new species, and that adjacent bare peat was colonized by herbs and bog-characteristic plants. These results were not tested for statistical significance. Over six years, translocated bog vegetation retained its overall bog-characteristic community (data reported as a graphical analysis). However, it did gain six additional plant species (before translocation: 15 species; six years after: 21 species) and abundance of fringed bog moss *Sphagnum fimbriatum* declined (in 15% of quadrats before translocation, but only 3% six years after). Bare peat between translocated strips was colonized by 28 plant species with 48% total vegetation cover, 21–31% grass/rush cover, 10–15% cover of heather *Calluna vulgaris* and 1–5% cover of five other bog-characteristic species. In 1991, sods (vegetation and 1 m of underlying peat) were cut from a blanket bog remnant. They were moved to eight 4 x 140 m trenches, dug in a site historically mined for coal. Dry peat was spread between the translocated strips. Plant species and vegetation cover were recorded in 1991 (before translocation) and 1997: in 100 quadrats (0.25 m^2) in six translocated strips, and in 90 quadrats (1 m^2) in three strips between.

A site comparison study in 2008–2009 in a fen in Ontario, Canada (2) reported that plots where surface peat was replaced developed plant cover and community composition intermediate between hummocks and hollows of an undisturbed plot. These results were not tested for statistical significance. After one year, *Sphagnum* moss cover was higher in peat-replacement plots (22–35%) than in undisturbed hollows (8–19%), but lower than on undisturbed hummocks (100%). The same was true for shrubs (peat-replacement: 15–20%; undisturbed hollows: 10%; undisturbed hummocks: 50%). For peat-replacement plots, data were not provided separately for hollows and hummocks. Overall community data were reported as a graphical analysis. In April 2008, 30-cm-thick blocks of peat and vegetation were replaced on a 12 x 12 m plot after removal of the underlying peat. An undisturbed plot 80 m away provided a comparison. From May to July 2009, vegetation cover was estimated in 6–18 quadrats/plot, distributed evenly across hummocks and hollows.

(1) Standen V. & Owen M.J. (1999) An evaluation of the use of translocated blanket bog vegetation for heathland restoration. *Applied Vegetation Science*, 2, 181–188.
(2) Wilhelm L.P., Morris P.J., Granath G. & Waddington J.M. (2015) Assessment of an integrated peat-harvesting and reclamation method: peatland-atmosphere carbon fluxes and vegetation recovery. *Wetlands Ecology and Management*, 23, 491–504.

4.2 Retain/create habitat corridors in areas of energy production or mining

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation, in habitat patches or within corridors, of retaining/creating habitat corridors in areas of energy production or mining.

Background

Habitat corridors are strips that link two larger habitat patches – in this case preventing peatland patches being separated by energy production or mining activities. By connecting the habitat patches, corridors could improve survival prospects of peatland plant populations. Seeds, pollen or vegetation fragments can more easily move between the habitat patches (perhaps carried by animals), maintaining populations and diversity in each (Damschen *et al.* 2006).

CAUTION: Habitat corridors can have negative effects. For example, corridors can allow diseases, non-native species and fire to spread between patches (Resasco *et al.* 2014).

Related interventions: maintaining/creating habitat corridors across service corridors (Section 5.3); rewetting, because peatlands may be drained directly for mining or dried out by drainage of sites nearby (Sections 8.1 and 14.7); habitat creation and restoration (Chapter 12).

Damschen E.I., Haddad N.M., Orrock J.L., Tewksbury J.J. & Levey D.J. (2006) Corridors increase plant species richness at large scales. *Science*, 313, 1284–1286.

Resasco J., Haddad N.M., Orrock J.L., Shoemaker D., Brudvig L., Damschen E.I., Tewksbury J.J. & Levy D.J. (2014) Landscape corridors can increase invasion by an exotic species and reduce diversity of native species. *Ecology*, 95, 2033–2039.

5. Threat: Transportation and service corridors

Background

This chapter addresses threats from long, narrow transport routes (e.g. roads, railways and canals) or service corridors (e.g. pipelines). These destroy the land on which they are built, but also impact the surviving habitat either side (Coffin 2007; Ryder *et al.* 2004). Transport routes and service corridors split the landscape into smaller fragments, can limit dispersal of native animals (and plant seeds or pollen they may carry) and can block flows of water into or out of peatlands (so they become too dry or too wet). Even 'floating roads' built without digging into the peat can affect water flows: they often subside over time as the underlying peat is compressed.

Related threats: residential and commercial development with a larger footprint (Chapter 2); human intrusions and disturbance, including damage from vehicles driving on peatlands (Chapter 7); invasive and problematic species, which may hitchhike along transport routes (Chapter 9); pollution from transport and service corridors e.g. air pollution, road salt, oil spills (Chapter 10). Related interventions: general habitat creation and restoration (Chapter 12); legal protection (Chapter 14).

Coffin A.W. (2007). From roadkill to road ecology: a review of the ecological effects of roads. *Journal of Transport Geography*, 15, 396–406.

Ryder A., Taylor D., Walters F. & Domeney R. (2004) Pipelines and peat: a review of peat formation, pipeline construction techniques and reinstatement options. Pages 582–601 in M. Sweeney (ed.) *Terrain and Geohazard Challenges Facing Onshore Oil and Gas Pipelines*. Conference Proceedings. London, UK.

Key messages

5.1 *Backfill trenches dug for pipelines* **0 studies**

We captured no evidence for the effect on peatland vegetation of backfilling pipeline trenches.

5.2 *Maintain/restore water flow across service corridors* **1 study**

Characteristic plants: One before-and-after study in a fen in the USA found that after restoring water inflow across a road, along with general rewetting, cover of wet peatland sedges increased whilst cover of grasses preferring drier conditions decreased.

5.3 *Retain/create habitat corridors across service corridors* **0 studies**

We captured no evidence for the effect on peatland vegetation, in habitat patches or within corridors, of retaining/creating habitat corridors across service corridors.

Interventions

5.1 Backfill trenches dug for pipelines

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of backfilling pipeline trenches.

Background

After pipelines have been dug in the surface of peatlands, peat could be replaced on top. This would provide the opportunity for peatland vegetation to redevelop.

Related interventions: rewetting, because peatlands may be drained to allow pipeline construction (Section 8.1); filling/blocking other types of ditches, trenches or gullies (Section 12.3); general habitat creation and restoration, to restore vegetation following replacement of peat (Chapter 12).

5.2 Maintain/restore water flow across service corridors Ⓑ Ⓕ Ⓢ

- **One study** examined the effect on peatland vegetation of restoring water flow across service corridors. The study was in a fen.
- **Characteristic plants (1 study):** One before-and-after study in a fen in the USA[1] found that following restoration of water inflow across a road (along with general rewetting), cover of wet peatland sedges increased whilst cover of grasses preferring drier conditions decreased.

Background

Peatlands may depend upon natural inflows or outflows of water, both above and below the surface, to maintain appropriate moisture and chemical conditions. Transportation or service corridors can block these water flows. This impact can be avoided by careful design e.g. building using permeable materials, or building raised service corridors (but note that *floating roads* built without digging into peat still block surface flow, and often sink over time so will block subsurface flow). Alternatively, water flows can be restored by building culverts, pipes or water diversions.

Related interventions: rewetting more generally (Section 8.1); restoring water level fluctuations (Section 8.4).

A before-and-after study in 2002–2004 in a degraded fen in California, USA (1) found that after water flow was restored across a road (along with general rewetting), cover of peatland-characteristic sedges increased whilst cover of grass species preferring drier conditions decreased. Cover of three sedge species characteristic of wet peatlands increased (two significantly or marginally so), from 12–15% before rewetting to 13–20% one year after. Cover of three grass species that prefer drier conditions decreased (two significantly), from 2–6% before rewetting to 1–5% one year after. In July 2003, 21 channels were created across a road that blocked surface water flow into the fen. At the same time, the main drainage ditch of the fen was dammed. The study does not distinguish between the effects of these interventions. Vegetation cover was estimated in July before (2002) and after (2004) intervention, in fifty-nine 10 m^2 plots.

(1) Patterson L. & Cooper D.J. (2007) The use of hydrologic and ecological indicators for the restoration of drainage ditches and water diversions in a mountain fen, Cascade Range, California. *Wetlands*, 27, 290–304.

5.3 Retain/create habitat corridors across service corridors Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation, in habitat patches or within corridors, of retaining/creating habitat corridors across service corridors.

Background

Habitat corridors are strips that link two larger habitat patches – in this case preventing peatland patches being separated by transportation or service corridors. By connecting the habitat patches, habitat corridors could improve survival prospects of peatland plant populations. Seeds, pollen or vegetation fragments can be moved along corridors (e.g. by animals), maintaining populations and diversity in each patch (Damschen *et al.* 2006).

CAUTION: Habitat corridors can have negative effects. For example, corridors can allow diseases, non-native species and fire to spread between patches (Resasco *et al.* 2014).

Related interventions: maintaining/creating habitat corridors in areas of energy production or mining (Section 4.2); habitat creation and restoration (Chapter 12).

Damschen E.I., Haddad N.M., Orrock J.L., Tewksbury J.J. & Levey D.J. (2006) Corridors increase plant species richness at large scales. *Science*, 313, 1284–1286.

Resasco J., Haddad N.M., Orrock J.L., Shoemaker D., Brudvig L., Damschen E.I., Tewksbury J.J. & Levy D.J. (1991) *Peat resources use in Canada: a national conservation issue*. Proceedings, International Peat Symposium. Duluth.

6. Threat: Biological resource use

Background

This chapter addresses threats from harvesting live, naturally occurring plants. This can be a major threat: 23% of South East Asian peat swamps have been damaged by logging, specifically the removal of valuable but long-lived (hence slow to recover) trees (Miettinen *et al.* 2016). Removing too much plant material from an area can be damaging – to individual plants or the whole population. The harvesting process can damage the physical structure of the bog too: by destroying hummocks and hollows, removing the living surface layer, and creating ruts as vehicles run over soft wet peat.

Numerous botanical resources are harvested from peatlands. Soft and absorbent *Sphagnum* moss is used as a substrate for growing plants, insulation, packaging, lining diapers and cleaning oil spills (Zegers *et al.* 2006). Fen grasses can be used for animal feed. Reeds are used for construction e.g. roof thatching (Schröder *et al.* 2015). Timber, fruits, latex and medicinal plants are amongst the products harvested from tropical peat swamp forests (Giesen 2015).

Related threats: agriculture and aquaculture i.e. harvesting resources in artificial environments (Chapter 3); peat excavation or mining (Chapter 4); drainage, which may be done to prepare land for harvest (Chapter 8). Related interventions: general habitat creation and restoration (Chapter 12); laws and agreements to encourage sustainable harvesting (Chapter 14).

Giesen W. (2015) Utilising non-timber forest products to conserve Indonesia's peat swamp forests and reduce carbon emissions. *Indonesian Journal of Natural History*, 3, 10–19.

Miettinen J., Shi C. & Liew S.C. (2016) Land cover distribution in the peatlands of Peninsular Malaysia, Sumatra and Borneo in 2015 with changes since 1990. *Global Ecology and Conservation*, 6, 67–78.

Schröder C., Dahms T., Paulitz J., Wichtmann W. & Wichmann S. (2003) Towards large-scale paludiculture: addressing the challenges of biomass harvesting in wet and rewetted peatlands. *Mires and Peat*, 16, Article 13.

Zegers G., Larraín J., Francisca Díaz M. & Armesto J. (2006) Impacto ecológico y social de la explotación de pomponales y turberas de *Sphagnum* en la Isla Grande de Chiloé (Ecological and social impact of the exploitation of mosses and *Sphagnum* bogs on the island of Chiloé ; in Spanish). *Revista Ambiente y Desarrollo*, 22, 28–34.

Key messages

6.1 *Reduce frequency of harvest* **0 studies**

We captured no evidence for the effect on peatland vegetation of reducing harvest frequency.

6.2 *Reduce intensity of harvest* **1 study**

Moss cover: One replicated, controlled study in a bog in New Zealand reported that *Sphagnum* moss cover was higher, three years after harvesting, when some *Sphagnum* was left in plots than when it was completely harvested.

6.3 *Use low impact harvesting techniques* **0 studies**

We captured no evidence for the effect on peatland vegetation of using low impact harvesting techniques.

6.4 *Use low impact vehicles for harvesting* **0 studies**

We captured no evidence for the effect on peatland vegetation of using specialized low impact vehicles for harvesting.

6.5 *Implement 'mosaic management' when harvesting wild biological resources* **0 studies**

We captured no evidence for the effect on peatland vegetation of implementing mosaic management when harvesting wild biological resources.

6.6 *Provide new technologies to reduce pressure on wild biological resources* **0 studies**

We captured no evidence for the effect on peatland vegetation of providing new technologies (e.g. fuel-efficient stoves) to reduce pressure on wild biological resources.

Interventions

6.1 Reduce frequency of harvest

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of reducing harvest frequency.

Background

Harvesting peatland vegetation less often (e.g. every two years instead of every year) will allow longer for the vegetation to recover from the disturbance, potentially growing taller and more densely. It might allow plants to mature and reproduce.

CAUTION: In some fens and fen meadows, regular harvesting is important to maintain the diversity of desirable peatland species (Middleton *et al*. 2006).

Related interventions: reduce intensity of harvest (Section 6.2); restrict vehicle use on peatlands (Section 7.1); allow sustainable use of peatlands (Section 14.6).

Middleton B., Holsten B. & van Diggelen R. (2006) Biodiversity management of fens and fen meadows by grazing, cutting and burning. *Applied Vegetation Science*, 9, 307–316.

6.2 Reduce intensity of harvest

Ⓑ Ⓕ Ⓢ

- **One study** examined the effect on peatland vegetation of reducing harvest intensity. The study was in a bog.
- **Moss cover (1 study):** One replicated, controlled study in a bog in New Zealand[1] reported that *Sphagnum* moss cover was higher, three years after harvesting, when some *Sphagnum* was left in plots than when it was completely harvested.

Background

Mechanical harvests can be intense, completely clearing vegetation from a peatland. Harvesting vegetation less intensely can increase its capacity to recover. Harvesting by hand is often less intense, or allows better control of intensity (Zegers *et al.* 2006). Harvesting a smaller area or removing fewer plants leaves a larger population of plants to grow or spread into harvested gaps. Removing less of each plant (e.g.

removing *Sphagnum* to a shallower depth) might avoid killing them and allow them to regrow (Díaz & Silva 2005). Low intensity harvests can often continue year after year.

CAUTION: In some fens and fen meadows, harvesting is important to maintain the diversity of desirable peatland species (Middleton *et al*. 2006).

Related interventions: reduce frequency of harvest (Section 6.1); allow sustainable use of peatlands (Section 14.6).

Díaz M.F. & Silva W. (2012) Improving harvesting techniques to ensure *Sphagnum* regeneration in Chilean peatlands. *Chilean Journal of Agricultural Research*, 72, 296–300.

Middleton B., Holsten B. & van Diggelen R. (2006) Biodiversity management of fens and fen meadows by grazing, cutting and burning. *Applied Vegetation Science*, 9, 307–316.

Zegers G., Larraín J., Francisca Díaz M. & Armesto J. (2006) Impacto ecológico y social de la explotación de pomponales y turberas de *Sphagnum* en la Isla Grande de Chiloé (Ecological and social impact of the exploitation of mosses and *Sphagnum* bogs on the island of Chiloé ; in Spanish). *Revista Ambiente y Desarrollo*, 22, 28–34.

A replicated, controlled study in a bog in New Zealand (1) reported that incompletely harvested plots regained *Sphagnum* moss cover more quickly than completely harvested plots. In plots where 30% of harvestable *Sphagnum* was left in place, *Sphagnum* cover was 90% after three years. In contrast, in plots from which all *Sphagnum* had been harvested, *Sphagnum* cover was only 50% after three years. No statistical tests were carried out and details of methods were not reported.

(1) Whinam J. & Buxton R.P. (1997) *Sphagnum* peatlands of Australasia: an assessment of harvesting sustainability. *Biological Conservation,* 82, 21–29.

6.3 Use low impact harvesting techniques

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of using low impact harvesting techniques.

Background

The impact of biological resource harvests could be reduced by better planning and/or using alternative techniques. For example, vehicles used for harvesting can compress, sink into and create ruts in wet peat soils. These impacts can be reduced by planning routes so the same ground is not repeatedly crossed, whilst ensuring vehicles are not overloaded and heavy (Schröder *et al.* 2015). Bails of vegetation could be rolled behind vehicles rather than carried on vehicles, spreading their weight over a larger area (Dubowski *et al.* 2013). In tropical forests, techniques to reduce logging impacts include directing the fall of felled trees, cutting lianas before felling (so they don't drag down other trees linked to the felled tree) and planning to keep disturbance from roads as small as possible (FAO 2004).

To be included as evidence, studies must have compared low impact harvesting techniques with alternative, traditional techniques.

Related interventions: reduce frequency of harvest (Section 6.1); reduce intensity of harvest (Section 6.2); restrict vehicle use on peatlands (Section 7.1).

Dubowski A.P., Zembrowski K., Rakowicz A., Palowski T., Weymann S. & Wojnilowicz L. (2013) Developing new-generation machinery for vegetation management on protected wetlands in Poland. *Mires and Peat*, 13, Article 11.

FAO (2004) *Reduced Impact Logging in Tropical Forests: Literature Synthesis, Analysis and Prototype Statistical Framework*. Forest Harvesting and Engineering Programme. Food and Agriculture Organization of the United Nations, Rome.

Schröder C., Dahms T., Paulitz J., Wichtmann W. & Wichmann S. (2003) Towards large-scale paludiculture: addressing the challenges of biomass harvesting in wet and rewetted peatlands. *Mires and Peat*, 16, Article 13.

6.4 Use low impact vehicles for harvesting

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of using specialized low impact vehicles for harvesting.

Background

Vehicles used to cut and transport peatland vegetation can be damaging to the peatland. They can compress, sink into and create ruts in the wet peat soils (Schröder *et al.* 2015). Using specialised tracked vehicles or hovercraft may reduce pressure on soils and mitigate some physical damage (Dubowski *et al.* 2013). To be included as evidence in this section, studies must have compared alternative low impact vehicles with traditional vehicles.

CAUTION: Even specially designed vehicles could alter the chemistry of the peatland by crushing vegetation and forcing it under water (Banaszuk et al. 2016).

Related interventions: restrict vehicle use on peatlands (Section 7.1); allow sustainable use of peatlands (Section 14.6).

Banaszuk P., Kamocki A. & Zarzecki R. (2016) Mowing with invasive machinery can affect chemistry and trophic state of a rheophilous mire. *Ecological Engineering*, 86, 31–38.

Dubowski A.P., Zembrowski K., Rakowicz A., Palowski T., Weymann S. & Wojnilowicz L. (2013) Developing new-generation machinery for vegetation management on protected wetlands in Poland. *Mires and Peat*, 13, Article 11.

Schröder C., Dahms T., Paulitz J., Wichtmann W. & Wichmann S. (2003) Towards large-scale paludiculture: addressing the challenges of biomass harvesting in wet and rewetted peatlands. *Mires and Peat*, 16, Article 13.

6.5 Implement 'mosaic management' when harvesting wild biological resources

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of implementing mosaic management when harvesting wild biological resources.

Background

Mosaic management involves managing neighbouring patches of land in different ways. For example, while some patches might be fully harvested, others might not be harvested in a given year. Patches may be harvested at different times within a given year. In this way, there are always parts of the peatland with older vegetation that could assist re-vegetation of harvested patches. This system could be implemented by individual landowners and/or at a larger scale across land owned by multiple people (Dicks *et al.* 2013).

Related intervention: implement mosaic management of farmland (Section 3.1).

Dicks L.V., Ashpole J.E., Dänhardt J., James K., Jönsson A., Randall N., Showler D.A., Smith R.K., Turpie S., Williams D. & Sutherland W.J. (2013) *Farmland Conservation: Evidence for the Effects of Interventions in Northern and Western Europe*. Pelagic Publishing, Exeter.

6.6 Provide new technologies to reduce pressure on wild biological resources

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of providing new technologies (e.g. fuel-efficient stoves) to reduce pressure on wild biological resources.

Background

Providing new, efficient technologies to people who live on or near peatlands, especially in developing countries, could reduce pressure on wild biological resources. More efficient equipment would use less of the natural resource, reducing the amount that needs to be harvested. For example, fuel-efficient stoves have been provided to reduce logging by inhabitants of Indonesian peat swamp forests (Rimba Raya 2017) and peat extraction from Andean bogs (BirdLife International 2013). The effects of these projects on peatlands were not quantitatively monitored. New technologies might also have health benefits e.g. less, or less harmful, smoke produced.

Related intervention: allow sustainable use of peatlands (Section 14.6).

BirdLife International (2013) Lake Junín: protection and sustainable use of High-Andean ecosystems. Available at http://www.birdlife.org/americas/news/lake-jun%C3%ADn-protection-and-sustainable-use-high-andean-ecosystems. Accessed 3 October 2017.

Rimba Raya (2017) Rimba Raya Biodiversity Reserve Project Overview. Available at http://rimba-raya.com/wp-content/uploads/Rimba-Raya-Project-Overview.pdf. Accessed 3 October 2017.

7. Threat: Human intrusions and disturbance

Background

The landscape, biodiversity, challenging physical conditions and isolation of peatlands make them attractive for a variety of non-consumptive uses (Bonn *et al.* 2009). However, activities of tourists, recreational users (including sporting events), land managers, scientists and the military can cause severe damage to peatlands. Pedestrians can trample peatland vegetation and erode fragile peat. Vehicles such as airboats can flatten vegetation and create channels that alter water flows (Racine *et al.* 1998). During war or military training, vehicle use, foot soldiers and bombing can destroy vegetation: in both temperate (Karofeld 1999; Rotherham & Handley 2013) and tropical peatlands (e.g. use of napalm in South East Asia).

Related threats: development of tourism infrastructure (Chapter 2 and Chapter 5); problematic species that might be introduced by visitors to peatlands (Chapter 9). Related interventions: general habitat creation and restoration (Chapter 12).

Bonn A., Rebane M. & Reid C. (2009) Ecosystem services: a new rationale for conservation of upland environments. Pages 448–474 in: A. Bonn, T. Allott, K. Hubacek & J. Stewart (eds.) *Drivers of Environmental Change in Uplands*. Routledge, London and New York.

Karofeld E. (1999) Effects of bombing and regeneration of plant cover in Kõnnu-Suursoo raised bog, North Estonia. *Wetlands Ecology and Management*, 6, 253–259.

Racine C.H., Walters J.C. & Jorgenson M.T. (1998) Airboat use and disturbance of floating mat fen wetlands in Interior Alaska. *Arctic*, 51, 371–377.

Rotherham I.D. & Handley C. (eds.) (2013) *War & Peat.* Wildtrack Publishing, Sheffield.

Key messages

7.1 *Restrict vehicle use on peatlands* **0 studies**

We captured no evidence for the effect on peatland vegetation of restricting vehicle use on peatlands.

7.2 *Physically exclude vehicles from peatlands* **1 study**

Vegetation structure: One replicated, paired, controlled, site comparison study in a floating fen in the USA reported that fencing off airboat trails allowed total and non-woody vegetation biomass to increase, up to levels recorded in undisturbed fen. Woody plant biomass did not recover.

Overall plant richness/diversity: The same study reported that fencing off airboat trails allowed overall plant diversity to increase, recovering to levels recorded in undisturbed fen.

7.3 *Restrict pedestrian access to peatlands* **0 studies**

We captured no evidence for the effect on peatland vegetation of restricting pedestrian access to peatlands.

7.4 *Physically exclude pedestrians from peatlands* **0 studies**

We captured no evidence for the effect on peatland vegetation of physically excluding pedestrians from peatlands.

7.5 *Install boardwalks/paths to prevent trampling* **0 studies**

We captured no evidence for the effect on peatland vegetation of installing boardwalks or paths to prevent trampling.

7.6 *Wear snowshoes to prevent trampling* **0 studies**

We captured no evidence for the effect on peatland vegetation of wearing snowshoes to prevent trampling.

7.7 *Adopt ecotourism principles/create an ecotourism site* **0 studies**

We captured no evidence for the effect on peatland vegetation of adopting ecotourism principles or creating an ecotourism site.

Interventions

7.1 Restrict vehicle use on peatlands

Ⓑ Ⓕ S

- We captured no evidence for the effect on peatland vegetation of restricting vehicle use on peatlands.

Background

Vehicles (e.g. cars, quad bikes, cycles and airboats) can directly damage peatland vegetation. They can also compress and rut soft, wet peat, affecting storage and flow of water which in turn affects peatland vegetation. To prevent this damage, vehicle use could be reduced by interventions such as legislation, limits on visitor numbers, voluntary codes, signage and/or ensuring official routes are well maintained.

Related interventions: use specialized low-impact vehicles to harvest wild biological resources on peatlands (Section 6.4); physically exclude vehicles from peatlands (Section 7.2); legally protect peatlands (Section 14.1).

7.2 Physically exclude vehicles from peatlands

Ⓑ Ⓕ S

- **One study** examined the effect on peatland vegetation of physically excluding vehicles from peatlands. The study was in a fen.
- **Vegetation structure (1 study):** One replicated, paired, controlled, site comparison study in a floating fen in the USA[1] reported that fencing off airboat trails allowed total and non-woody vegetation biomass to increase, recovering to levels recorded in undisturbed fen. Woody plant biomass did not recover.
- **Overall plant richness/diversity (1 study):** The same study[1] reported that fencing off airboat trails allowed overall plant diversity to increase, recovering to levels recorded in undisturbed fen.

Background

Vehicles (e.g. cars, quad bikes, cycles and airboats) can directly damage peatland vegetation. They can also compress and rut soft, wet peat, affecting storage and flow of water which in turn affects peatland vegetation. Vehicles could be physically excluded from pristine peatlands to prevent damage, or from damaged peatlands to let them recover. Physical barriers could be fences, fallen trees or areas of water/wet ground.

Related intervention: restrict vehicle use on peatlands, using non-physical means such as signs or voluntary codes (Section 7.1).

A replicated, paired, controlled, site comparison study in 2002–2005 in a floating fen in Alaska, USA (1) reported that plots fenced off from airboats developed greater plant diversity and non-woody plant biomass than exposed plots, similar to natural fen vegetation. Comparisons with exposed plots were not tested for statistical significance. After three years, plant diversity in fenced plots was higher than in exposed plots, and not significantly different from diversity in natural plots (data reported as a diversity index). The same was true for total above-ground plant biomass (fenced: 149; exposed: 49; natural: 242 g/m^2), sedge biomass (fenced: 92; exposed: 24; natural: 83 g/m^2) and forb biomass (fenced: 50; exposed: 24; natural: 47 g/m^2). In contrast, woody plant biomass had not recovered in fenced plots (fenced: 5; exposed: 0; natural: 110 g/m^2). Three months after fencing, all measures were no different, or lower, in fenced plots compared to exposed plots. In March 2002, eight sets of three 3.25 m^2 plots were established. In each set, one plot was in natural fen vegetation and two were in airboat trails. Airboats were excluded from one of these plots by erecting log tripods. In summer 2002–2005, vegetation was cut from one 25 x 25 cm quadrat/plot then identified, dried and weighed.

(1) Zacheis, A. & Doran, K. (2009) Resistance and resilience of floating mat fens in interior Alaska following airboat disturbance. *Wetlands*, 29, 236–247.

7.3 Restrict pedestrian access to peatlands

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of restricting pedestrian access to peatlands.

Background

Walking on peatlands can damage the vegetation and compress or erode the peat (e.g. Slater & Agnew 1977). Pedestrians are a particular problem when they repeatedly walk on the same area of peatland e.g. in popular tourist areas, or when scientists make repeat visits to sample plots. To prevent this damage, pedestrian access to peatlands could be reduced by interventions such as legislation, limits on visitor numbers, voluntary codes, signage and/or ensuring official paths are well maintained.

Related interventions: physically exclude pedestrians from peatlands (Section 7.4); install boardwalks to prevent trampling (Section 7.5).

Slater F.M. & Agnew A.D.Q. (1977) Observations on a peat bog's ability to withstand increasing public pressure. *Biological Conservation*, 11, 21–27.

7.4 Physically exclude pedestrians from peatlands

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of physically excluding pedestrians from peatlands.

Background

Walking on peatlands can damage the vegetation and compress or erode the peat (e.g. Slater & Agnew 1977). This is a particular problem when the same area of peatland is repeatedly crossed e.g. in popular hiking areas, in tourist sites/nature reserves, or when scientists make repeat visits to sample plots. Pedestrians could be physically excluded from pristine peatlands to prevent damage, or from damaged peatlands to let them recover. Physical barriers could be fences, fallen trees or water/wet ground.

Related intervention: restrict pedestrian access to peatlands, using non-physical means such as signs or voluntary codes (Section 7.3).

Slater F.M. & Agnew A.D.Q. (1977) Observations on a peat bog's ability to withstand increasing public pressure. *Biological Conservation*, 11, 21–27.

7.5 Install boardwalks/paths to prevent trampling Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of installing boardwalks or paths to prevent trampling.

Background

Walking on peatlands can damage the vegetation and compress or erode the peat. Pedestrians are a particular problem when they repeatedly walk on the same area of peatland e.g. in popular hiking areas, in tourist sites/nature reserves, or when scientists repeatedly visit sample plots (Edwards 1977). Installing boardwalks or designated paths can prevent physical contact with the peatland – assuming people stay on the boardwalks.

CAUTION: Preservatives leaching from timber may damage peatland vegetation. Boardwalks will also shade and kill the vegetation beneath. Paths can compress peat and alter water flow patterns, above and below the peatland surface.

Related interventions: restrict pedestrian access to peatlands (Section 7.3); physically exclude pedestrians from peatlands (Section 7.4).

Edwards I.J. (1977) The ecological impact of pedestrian traffic on alpine vegetation in Kosciusko National Park. *Australian Forestry*, 40, 108–120.

7.6 Wear snowshoes to prevent trampling Ⓑ Ⓕ S

- We captured no evidence for the effect on peatland vegetation of wearing snowshoes to prevent trampling.

Background

Walking on peatlands can damage the vegetation and compress or erode the peat. Pedestrians are a particular problem when they repeatedly walk on the same area of peatland e.g. in popular tourist areas or when scientists make repeat visits to sample plots (Edwards 1977). Wearing snowshoes could spread the weight of people walking on peatlands and minimise trampling impacts.

Related interventions: use specialized low impact vehicles for harvest (Section 6.4); restrict pedestrian access to peatlands (Section 7.3).

Edwards I.J. (1977) The ecological impact of pedestrian traffic on alpine vegetation in Kosciusko National Park. *Australian Forestry*, 40, 108–120.

7.7 Adopt ecotourism principles/create an ecotourism site Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of adopting ecotourism principles or creating an ecotourism site.

Background

Tourists may visit peatlands for many reasons e.g. hiking, biking, skiing, viewing wild animals, photography and relaxation. Tourist visits could be managed with conservation in mind: minimizing damage from tourist activities, educating both staff and visitors, and providing a financial incentive to conserve natural peatland (The International Ecotourism Society 2017). These principles could be adopted by existing tourist sites, or new ecotourism sites could be created.

Related interventions: limit damage that may arise from tourist activities (Sections 7.1–7.6); legally protect peatlands (Section 14.1); raise awareness amongst the public, including about how to avoid damaging peatlands (Section 15.1).

The International Ecotourism Society (2017) *What is Ecotourism?* Available at http://www.ecotourism.org/what-is-ecotourism. Accessed 1 August 2017.

8. Threat: Natural system modifications

Background

This chapter addresses threats from changes in management of natural or semi-natural peatlands – including changes in water levels, traditional management and fire regimes. Such changes are often involved in peatland degradation, in both temperate and tropical regions, and can have large impacts.

Water levels within peatlands may be managed to benefit humans. In particular, peatlands are drained to grow crops, harvest peat or vegetation, and build roads. Drained peatlands cannot support bog, fen or swamp vegetation: they are too dry and the peat chemistry can be unsuitable. Stabilizing water levels, even if the average water level stays the same, can also affect soil chemistry (Lamers *et al.* 2002). Finally, reservoirs built to generate power, supply water or for recreation can cause large scale, deep flooding of peatlands, eradicating their characteristic plant communities.

Regular disturbance such as burning, mowing or grazing may maintain desirable semi-natural peatland vegetation, especially in fen meadows (Broads Authority 2010). Disturbances can clear dominant reeds/shrubs/trees, create space for other plants to grow and prevent a build-up of nutrients. Consequently, they can maintain habitat structure and species diversity (Wiegers 1992). If historical disturbance or management regimes are stopped, desirable vegetation structure and diversity may be lost. In such cases, conservationists may want to maintain disturbance, restore disturbance, or use an intervention that compensates for a loss of disturbance.

However, disturbances that are too frequent or intense can damage, or change, peatland plant communities (e.g. Page *et al.* 2009; Worrall *et al.* 2010). This chapter also considers interventions to prevent excess disturbance from fire. Most peatlands do not, naturally, burn very often (Lindsay *et al*. 2011).

Related threats: conversion of land to artificial agricultural systems, rather than changing management within natural or semi-natural systems (Chapter 3). Related interventions: interventions from this chapter used to control problematic species, but where they have not benefited from a change in the historical disturbance regime (Chapter 9); general habitat creation and restoration (Chapter 12); education to prevent wild fire (Chapter 15).

Broads Authority (2010) *New Opportunities for the Sustainable Management of Fens: Reed Pelleting, Composting and the Productive Use of Fen Harvests.* Broads Authority Research Report.

Lamers L.P., Smolders A.J.P. & Roelofs J.G.M. (2002) The restoration of fens in the Netherlands. *Hydrobiologia*, 478, 107–130.

Lindsay R., Birnie R. & Clough J. (2011) *Burning*. IUCN UK Peatland Programme Briefing Note No. 8.

Page S., Hoscilo A., Langner A., Tansey K., Siegert F., Limin S. & Rieley J. (2009) Tropical peatland fires in Southeast Asia. Pages 263–287 in: Cochrane M. (ed.) *Tropical Fire Ecology*. Springer, Berlin.

Poulin M., Rochefort L., Pellerin S. & Thibault J. (2004) Threats and protection for peatlands in Eastern Canada. *Géocarrefour*, 79, 331–344.

Wiegers J. (1992) Carr vegetation: plant communities and succession of the dominant tree species. Pages 361–396 in: J.T.A. Verhoeven (ed.) *Fens and Bogs in the Netherlands: Vegetation, History, Nutrient Dynamics and Conservation*. Kluwer Academic Publishers, Dordecht.

Worrall F., Clay G.D., Marrs R. & Reed M.S. (2010) *Impact of Burning Management on Peatlands*. IUCN UK Peatland Programme Scientific Review.

Key messages

Modified water management

8.1 *Rewet peatland (raise water table)* **36 studies**

Plant community composition: Ten of thirteen studies reported that rewetting affected the overall plant community composition. Six before-and-after studies (four also replicated) in peatlands in Finland, Hungary, Sweden, Poland and Germany reported development of wetland- or peatland-characteristic communities following rewetting. One replicated, paired, controlled study in the Czech Republic found differences between rewetted and drained parts of a bog. Three site comparison studies in Finland and Canada reported differences between rewetted and natural peatlands. In contrast, three replicated studies in peatlands in the UK and fens in Germany reported that rewetting typically had no effect, or insignificant effects, on the plant community.

Characteristic plants: Five studies (including one replicated site comparison) in peatlands in Canada, the UK, China and Poland reported that rewetting, sometimes along with other interventions, increased the abundance of wetland- or peatland-characteristic plants. Two replicated site comparison studies, in fens and fen meadows in Europe, found that rewetting reduced the number of fen-characteristic plant species. Two studies (one replicated, paired, controlled, before-and-after) in fens in Sweden reported that rewetting had no effect on cover of fen-characteristic plants.

Moss cover: Twelve studies (two replicated, paired, controlled) in bogs, fens or other peatlands in Europe and Canada reported that rewetting, sometimes along with other interventions, increased *Sphagnum* moss cover or abundance. However two replicated studies, in bogs in Latvia and forested fens in Finland, reported that rewetting had no effect on *Sphagnum* cover. Five studies (one paired, controlled, before-and-after) in bogs and fens in Finland, Sweden and Canada reported no effect of rewetting on non-*Sphagnum* mosses/lichens. However two controlled studies, in bogs in Ireland and the UK, reported that rewetting reduced cover of non-*Sphagnum* mosses or bryophytes. Of two studies that compared rewetted and natural peatlands, one in Finland reported similar moss cover in both, but one in Canada reported that a rewetted bog had lower moss cover than target peatlands.

Herb cover: Twenty-one studies (four replicated, paired, controlled) reported that rewetting, sometimes along with other interventions, increased cover of at least one group of herbs: sedges in 13 of 15 studies, cottongrass in eight of nine studies and reeds/rushes in five of seven studies. The studies were in bogs, fens or other peatlands in Europe, North America and China. Of four before-and-after studies in peatlands in the UK and Sweden, three reported that rewetting reduced purple moor grass cover but one reported no effect. One replicated site comparison study, in forested fens in Finland, reported that rewetting had no effect on total herb cover. Two site comparison studies in Europe reported that rewetted peatlands had greater herb cover (total or sedges/rushes) than natural peatlands.

Tree/shrub cover: Ten studies (two paired and controlled) in peatlands in Finland, the UK, Germany, Latvia and Canada reported that rewetting typically reduced or had no effect on tree and/or shrub cover. Two before-and-after studies in fens in Sweden and Germany reported that tree/shrub cover increased following rewetting. One before-and-after study in a bog in the UK reported mixed effects of rewetting on different tree/shrub species.

Overall vegetation cover: Of four before-and-after studies (including three controlled), two in bogs in Ireland and Sweden reported that rewetting increased overall vegetation cover. One study in a fen in New Zealand reported that rewetting reduced vegetation cover. One study in a peatland in Finland reported no effect.

Overall plant richness/diversity: Six studies (including one replicated, paired, controlled, before-and-after) in Sweden, Germany and the UK reported that rewetting increased total plant species richness or diversity in peatlands. However, five studies found no effect: in bogs in the Czech Republic and Latvia, fens in Sweden and Germany, and forested fens in Finland. One study in fen meadows in the Netherlands found scale-dependent effects. One paired, controlled, before-and-after study in a peatland in Finland reported that rewetting reduced plant diversity. Of four studies that compared rewetted and natural peatlands, two in Finland and Germany reported lower species richness in rewetted peatlands, one in Sweden found higher species richness in rewetted fens, and one in Europe found similar richness in rewetted and natural fens.

Growth: One replicated site comparison study, in forested fens in Finland, found that rewetting increased *Sphagnum* moss growth to natural levels.

8.2 *Irrigate peatland* **2 studies**

Vegetation cover: One replicated, paired, controlled, before-and-after study in a bog in Canada found that irrigation increased the number of *Sphagnum* moss shoots present after one growing season, but had no effect after two. One before-and-after study in Germany reported that an irrigated fen was colonized by wetland- and fen-characteristic herbs, whilst cover of dryland grasses decreased.

8.3 *Reduce water level of flooded peatlands* **0 studies**

We captured no evidence for the effect on peatland vegetation of reducing the water level in unnaturally flooded peatlands.

8.4 *Restore natural water level fluctuations* **0 studies**

We captured no evidence for the effect on peatland vegetation of restoring natural water level fluctuations *per se*.

Modified vegetation management

8.5 *Cut/mow herbaceous plants to maintain or restore disturbance* **14 studies**

Plant community composition: Six replicated studies in fens and fen meadows in the UK, Belgium, Germany and the Czech Republic reported that mowing altered the overall plant community composition (vs no mowing, before mowing or grazing). One site comparison study in Poland reported that mowing a degraded fen, along with other interventions, made the plant community more similar to target fen meadow vegetation.

Characteristic plants: Four studies (including one replicated, paired, controlled, before-and-after) in fens and fen meadows in Switzerland, Germany, the Czech Republic and Poland found that cutting/mowing increased cover of fen- or wet meadow-characteristic plants. One replicated before-and-after study, in fens in the UK, found that a single mow typically had no effect on cover of fen-characteristic plants. In Poland and the UK, the effect of mowing was not separated from the effects of other interventions.

Moss cover: Four replicated, paired studies (three also controlled) in fens and fen meadows in Belgium, Switzerland and the Czech Republic found that mowing increased total moss or bryophyte cover. Two replicated studies (one also controlled) in fens in Poland and the UK found that a single mow typically had no effect on bryophyte cover (total or hollow-adapted mosses).

Herb cover: Six replicated studies (three also randomized and controlled) in fens and fen meadows in Belgium, Germany, Poland and the UK found that mowing reduced cover or abundance of at least one group of herbs (including bindweed, reeds, sedges, purple moor grass and grass-like plants overall). One before-and-after study in a fen in Poland found that mowing, along with other interventions, increased sedge cover. One replicated, randomized, paired, controlled study in fen meadows in Switzerland found that mowing had no effect on overall herb cover.

Shrub cover: Of three replicated studies in fens, two in the UK found that a single mow, sometimes along with other interventions, reduced shrub cover. The other study, in Poland, found that a single mow had no effect on shrub cover.

Vegetation structure: In the following studies, vegetation structure was measured 6–12 months after the most recent cut/mow. Three replicated studies in fens in Poland and the UK reported that a single mow, sometimes along with other interventions, had no (or no consistent) effect on vegetation height. One replicated, paired, site comparison study in fen meadows in Switzerland found that mowing reduced vegetation height. Three studies in fen meadows in Switzerland, Poland and Italy found mixed effects of mowing on vegetation biomass (total, sedge/rush, moss or common reed). One replicated, paired, site comparison study in Germany reported that vegetation structure was similar in mown and grazed fen meadows.

Overall plant richness/diversity: Eight studies in fens and fen meadows in the UK, Belgium, Switzerland, Germany, the Czech Republic and Poland found that mowing/cutting increased plant species richness (vs no mowing, before mowing or grazing). Three studies (two replicated, randomized, paired, controlled) in fens in Poland and the UK found that a single mow, sometimes along with other interventions, typically had no effect on plant species richness and/or diversity.

8.6 *Remove plant litter to maintain or restore disturbance* — 2 studies

Plant community composition: Two studies (including one replicated, paired, controlled, before-and-after) in a fen meadow in Germany and a fen in Czech Republic found that removing plant litter did not affect plant community composition.

Vegetation cover: One replicated, paired, controlled, before-and-after study in a fen in the Czech Republic found that removing litter did not affect bryophyte or moor grass cover.

Overall plant richness/diversity: Of two replicated, controlled studies, one (also randomized) in a fen meadow in Germany reported that removing plant litter increased plant species richness and diversity. The other study (also paired and before-and-after) in a fen in the Czech Republic found that removing litter did not affect vascular plant diversity.

8.7 *Cut large trees/shrubs to maintain or restore disturbance* — 2 studies

Plant community composition: One study in a fen in Poland found that where shrubs were removed, along with other interventions, the plant community composition became more like a target fen meadow over time.

Characteristic plants: One study in a fen in Poland found that where shrubs were removed, along with other interventions, the abundance of fen meadow plant species increased over time.

Vegetation cover: One replicated, paired, controlled study in a forested fen in the USA found that cutting and removing trees increased herb cover, but did not affect shrub cover.

Vegetation structure: One replicated, paired, controlled study in a forested fen in the USA found that cutting and removing trees increased herb biomass and height.

8.8 *Use grazing to maintain or restore disturbance* — 4 studies

Plant community composition: One replicated, paired, site comparison study in Germany found that the overall plant community composition differed between grazed and mown fen meadows.

Characteristic plants: One replicated, paired, controlled study in Germany reported that the abundance of bog/fen-characteristic plants was similar in grazed and ungrazed fen meadows. One replicated before-and-after study in a fen in the UK reported that grazing did not affect cover of fen-characteristic mosses. One replicated, paired, site comparison study in Germany found that grazed fen meadows contained fewer fen-characteristic plant species than mown meadows.

Herb cover: Two before-and-after studies in fens in the UK reported that grazing increased cover of some herb groups (cottongrasses, sedges or all grass-like plants). One of the studies found that grazing reduced purple moor grass cover, but the other found that grazing typically had no effect.

Moss cover: One replicated before-and-after study, in a fen in the UK, reported that cover of fen-characteristic mosses did not change after grazers were introduced. One controlled, before-and-after study in a fen in the UK found that grazing reduced *Sphagnum* moss cover.

Tree/shrub cover: Of two before-and-after studies in fens in the UK, one found that grazing reduced shrub cover but the other found that grazing typically had no effect on shrub cover.

Overall plant richness/diversity: Of two before-and-after studies in fens in the UK, one (also controlled) reported that grazing increased plant species richness but the other (also replicated) found that grazing had no effect. One replicated, paired, site comparison study in Germany found that grazed fen meadows contained fewer plant species than mown meadows.

8.9 *Use prescribed fire to maintain or restore disturbance* **3 studies**

Characteristic plants: One replicated before-and-after study in a fen in the UK reported that burning, along with other interventions, did not affect cover of fen-characteristic mosses or herbs.

Herb cover: One replicated, controlled study in a fen in the USA reported that burning reduced forb cover and increased sedge/rush cover, but had no effect on grass cover. One replicated before-and-after study in a fen in the UK reported that burning, along with other interventions, reduced grass/sedge/rush cover.

Tree/shrub cover: Two replicated studies in fens in the USA and the UK reported that burning, sometimes along with other interventions, reduced tree/shrub cover.

Overall plant richness/diversity: Two replicated, controlled studies in a fen in the USA and a bog in New Zealand found that burning increased plant species richness or diversity. However, one replicated before-and-after study in a fen in the UK reported that burning, along with other interventions, typically had no effect on plant species richness and diversity.

Modified wild fire regime

8.10 *Thin vegetation to prevent wild fires* **0 studies**

We captured no evidence for the effect on peatland vegetation of thinning vegetation to prevent wild fires.

8.11 *Rewet peat to prevent wild fires* **0 studies**

We captured no evidence for the effect on peatland vegetation of rewetting peat to prevent wild fires.

8.12 *Build fire breaks* **0 studies**

We captured no evidence for the effect on peatland vegetation of building fire breaks.

8.13 *Adopt zero burning policies near peatlands* **0 studies**

We captured no evidence for the effect on peatland vegetation of adopting zero burning policies near peatlands.

Interventions: Modified water management

8.1 Rewet peatland (raise water table) Ⓑ Ⓕ Ⓢ

- **Thirty-six studies** examined the effect of rewetting (without planting) on peatland vegetation. Fifteen studies were in bogs[1,3,4,5,11a,11b,14,15,16,20,23,24,27,33,34] (two[4,34] being restored as fens). Fifteen studies were in fens or fen meadows[6,7,8,9,10,12,17,19,21,22,25,28,29,30,31] (two[28,30] were naturally forested). Six studies were in general or unspecified peatlands[2,13,18a,18b,26,32]. Some studies were based on the same experimental set-up or sites as each other: two studies in Germany[14,16], three studies in Sweden[12,17,21], two studies in west Finland[2,13] and two studies in south Finland[28,30].

- **Plant community composition (13 studies):** Six before-and-after studies (four also replicated) in peatlands in Finland[2], Hungary[6], Sweden[12,33], Poland[22] and Germany[25] reported changes in the overall plant community composition following rewetting. Typically, drier grassland communities were replaced by more wetland- or peatland-characteristic communities. One replicated, paired, controlled study in a bog in the Czech Republic[5] found that rewetted plots developed a different plant community to drained plots. Three site comparison studies in Finland[13,28] and Canada[34] reported that rewetted peatlands contained a different plant community to natural peatlands. Three replicated studies in peatlands in the UK[26,32] and fens in Germany[29] reported that rewetting typically had no effect, or insignificant effects, on the plant community.
- **Characteristic plants (11 studies):** Five studies (including one replicated site comparison) in peatlands in Canada[4], the UK[15], China[18a,18b] and Poland[22] reported that rewetting (sometimes[22] along with other interventions) increased the abundance of wetland- or peatland-characteristic plants. Two replicated site comparison studies in fens and fen meadows in Europe[9,19] found that rewetting reduced the number of fen-characteristic plant species. Two studies (one replicated, paired, controlled, before-and-after) in fens in Sweden[10,17] reported that rewetting had no effect on cover of fen-characteristic plants. Two before-and-after studies in fens in the USA[7] and New Zealand[8] reported that upland plant cover decreased following rewetting.
- **Moss cover (19 studies):** Twelve studies (five replicated, two also paired and controlled) in the UK[1,32], Ireland[3], Germany[14,16], Sweden[10,12,17], Latvia[20,24], Canada[27] and Spain[31] reported that rewetting bogs, fens or other peatlands (sometimes[1,31] along with other interventions) increased *Sphagnum* moss cover or abundance. Three of these studies[14,27,32] reported mixed responses by species. Two additional replicated studies, in bogs in Latvia[23] and forested fens in Finland[28], reported that rewetting had no effect on *Sphagnum* cover. Five studies (one paired, controlled, before-and-after) in Finland[2,28], Sweden[10,33] and Canada[27] reported that rewetting bogs or fens had no effect on cover of non-*Sphagnum* mosses (or mosses/lichens[2]). However, two controlled studies in bogs in Ireland[3] and the UK[11a] reported that rewetting reduced cover of non-*Sphagnum* mosses[11a] or bryophytes[3]. One site comparison study in Finland[13] reported that a rewetted peatland had similar moss cover (*Sphagnum* and total) to a natural peatland, but another site comparison study in Canada[34] reported that a rewetted bog had lower moss cover (*Sphagnum* and other) than nearby target peatlands.
- **Herb cover (25 studies):** Twenty-one studies (including four replicated, paired, controlled) reported that rewetting (sometimes[1,22,31] along with other interventions) increased cover of at least one group of herbs. These studies were in bogs, fens or other peatlands in the UK[1,11a,11b,26], Finland[2], Ireland[3], the Czech Republic[5], the USA[7], the Netherlands[9], Sweden[12,17,33], Germany[14,16,25], China[18a,18b], Latvia[20], Poland[22], Canada[27] and Spain[31]. Specifically, rewetting increased sedge cover in 13 of 15 studies[1,2,7,9,12,13,14,16,17,19,22,25,33], increased cottongrass cover in eight of nine studies[1,2,3,11a,13,16,20,33], and increased reed/rush cover in five of seven studies[14,19,25,31,33]. Three[1,10,12] of four before-and-after studies in peatlands in the UK and Sweden reported that rewetting reduced purple moor grass cover; the other study[26] reported no effect. One replicated site comparison study in forested fens in Finland[28] reported that rewetting had no effect on total herb cover. Two site comparison studies in Europe[13,19] reported greater herb cover in rewetted than natural peatlands (overall[19] and sedges/rushes[13,19], but not forbs[13]).
- **Tree/shrub cover (13 studies):** Ten studies (including two paired and controlled) in peatlands in Finland[2,13,28], the UK[11b], Germany[14,16], Latvia[20,23,24] and Canada[27] reported that rewetting typically reduced[2,13,14,16,20,24,27] or had no effect[11b,16,20,23,27,28] on tree and/or shrub cover. Two before-and-after studies in fens in Sweden[10] and Germany[25] reported that rewetting increased tree/shrub

cover. One before-and-after study in a bog in the UK[1] reported mixed effects of rewetting on different tree/shrub species.

- **Overall vegetation cover (4 studies):** Of four before-and-after studies (three also controlled) that examined the effect of rewetting on overall vegetation cover, two in bogs in Ireland[3] and Sweden[33] reported that rewetting increased it. One study in a fen in New Zealand[8] reported that rewetting reduced vegetation cover. One study in a peatland in Finland[2] reported no effect.
- **Overall plant richness/diversity (14 studies):** Six studies (including one replicated, paired, controlled, before-and-after) in Sweden[10,21,33], Germany[14,16] and the UK[32] reported that rewetting increased total plant species richness or diversity in bogs, fens or other peatlands. However, five studies found no effect: in bogs in the Czech Republic[5] and Latvia[20], fens in Sweden[17] and Germany[29], and forested fens in Finland[28]. One study in fen meadows in the Netherlands[9] found scale-dependent effects. One paired, controlled, before-and-after study in a peatland in Finland[2] reported that rewetting reduced plant diversity. Of four studies that compared rewetted and natural peatlands, two in Finland[13] and Germany[29] reported lower species richness in rewetted peatlands, one in Sweden[21] found higher species richness in rewetted fens, and one in Europe[19] found similar richness in rewetted and natural fens.
- **Growth (1 study):** One replicated site comparison study in forested fens in Finland[30] found that rewetting increased *Sphagnum* moss growth to natural levels.

Background

Peatlands may be drained for activities such as crop cultivation, livestock grazing, natural resource harvesting, road building or urban development. Peatland drainage may also be unintentional e.g. extracting drinking water from below ground lowers the water table over a large area. Drained peat can be too dry and chemically unsuitable for peatland plants (Lamers *et al.* 2002). Raising the water table will rewet the surface peat, creating more suitable conditions for recolonization by peatland plants and less suitable conditions for other species (Money & Wheeler 1999; Ritzema *et al.* 2014). It may be necessary to rewet the area around a peatland too (creating a 'hydrological buffer zone') to prevent water simply draining away from the peatland.

A range of techniques may be used to raise the water table in peatlands e.g. blocking drainage ditches or gullies (using peat, rocks, plastic dams or wooden dams), planting flood-resistant vegetation in ditches to slow water flow, blocking underground channels or peat pipes, building raised embankments or berms (elongated mounds of peat or rows of straw bales) to retain water, inserting dams (e.g. straw bales) below the peat surface to slow subsurface drainage, switching off drainage pumps, or restoring inflows. These interventions are all considered in this section.

CAUTION: Deep flooding is generally not desirable when restoring peatland vegetation. The water table should be raised to anywhere from just below the peat surface to a few centimetres above, depending on the site. Also, rewetting may increase emissions of greenhouse gasses such as methane (Abdalla et al. 2016).

Related interventions: interventions to address residential and commercial development (Chapter 2), agriculture (Chapter 3) and peat mining (Chapter 4) which are often accompanied by drainage; irrigate peatland surface (Section 8.2); restoration using multiple interventions, often including rewetting (Section 12.1); fill/block drainage ditches and the effect of vegetation growing within them (Section 12.3); reprofile peatland, or remove peat, bringing the peatland surface closer to the water table (Sections 12.5 and 12.7); rewetting, irrigation and reprofiling to complement planting (Sections 13.7–13.9).

Abdalla M., Hastings A., Truu J., Espenberg M. & Mander Ü. (2016) Emissions of methane from northern peatlands: a review of management impacts and implications for future management options. *Ecology and Evolution*, 6, 7080–7102.

Lamers L.P., Smolders A.J.P. & Roelofs J.G.M. (2002) The restoration of fens in the Netherlands. *Hydrobiologia*, 478, 107–130.

Money R.P. & Wheeler B.D. (1999) Some critical questions concerning the restorability of raised bogs. *Applied Vegetation Science*, 2, 107–116.

Ritzema H., Limin S., Kusin K., Jauhiainen J. & Wösten H. (2014) Canal blocking strategies for hydrological restoration of degraded tropical peatlands in Central Kalimantan, Indonesia. *Catena*, 114, 11–20.

A before-and-after study in 1972–1987 in a historically mined raised bog in England, UK (1) reported that after rewetting (and diversion of polluted inflow), cover of *Sphagnum* moss, white sedge *Carex curta* and cottongrasses *Eriophorum* spp. increased, but cover of purple moor grass *Molinia caerulea* decreased. No statistical tests were carried out. *Sphagnum* was found in 7% of quadrats before intervention but 27% after, white sedge in 0.0% before but 0.8% after, and cottongrasses in 1.1% before but 1.5–1.7% after. In contrast, purple moor grass occurred in 100% of quadrats before intervention but only 74% after. Eighteen other herb, shrub and tree species showed variable responses (see original paper). In 1974, a drained bog was rewetted (surface partially waterlogged) by blocking its water outflow. At the same time, polluted inflow from adjacent farms was diverted off site. The study does not distinguish between the effects of these interventions. Vegetation cover was recorded before (1972–1973) and after (1987) intervention, as presence/absence of species in 8,945 contiguous 4 m^2 quadrats covering the whole site.

A paired, controlled, before-and-after study in 1994–1998 in a historically mined peatland in Finland (2) reported that rewetted plots developed a different plant community to drained plots with lower plant diversity and shrub cover, but similar moss/lichen and total vegetation cover. These results were not tested for statistical significance. Over four years, rewetted plots developed more peatland-characteristic plant communities than drained plots (data reported as a graphical analysis). After four years, plant diversity was lower in rewetted plots than drained plots (data reported as a diversity index). In rewetted plots, shrub cover was 0% (vs 6% in drained plots), moss/lichen cover 13–20% (drained: 19–20%), and total vegetation cover 40–60% (drained: 40–45%). Cottongrass *Eriophorum vaginatum* cover was 20–34% (vs 11–18% before rewetting) and sedge *Carex* spp. cover was 12% (vs 4% before rewetting). Before intervention, plots to be rewetted or drained had similar plant diversity, shrub cover (0–11%), moss/lichen cover (2–24%) and total vegetation cover (15–50%). In autumn 1994, four plots were established within one drained peat field. Two plots were rewetted by blocking drainage ditches with peat dams and digging a new input ditch. Two plots remained drained. Every summer between 1994 and 1998, cover of every plant species was estimated in twelve 2 m^2 quadrats/plot. The water table was, on average, within 18 cm of the peat surface in rewetted plots, (vs drained plots: 4–31 cm below). This study used the same rewetted peatland as (13).

A controlled before-and-after study in 1997–1999 in a historically mined blanket bog in Ireland (3) reported that after rewetting, cover of total vegetation, algae, *Sphagnum* moss and common cottongrass *Eriophorum angustifolium* all increased, but cover of other bryophytes decreased. No statistical tests were carried out. Total vegetation cover was 40% before rewetting and 93% two years after. Cover of algae was 10% before and 92% after, *Sphagnum* <1% before and 31% after, other bryophytes 12% before and 1% after, and cottongrass 18% before and 24% after. In control plots that remained drained, cover values remained stable over time (total: 54–56%; algae: 21%; *Sphagnum*: 0%; other bryophytes: 26%; cottongrass: 31–32%). In August 1997, a 1 m high peat ridge was built around a drained bog to retain water (water table raised to 10–42 cm above peat surface). An adjacent bog remained drained for comparison (water table 0–12 cm below surface). In August 1997 (before intervention) and 1999, vegetation cover was estimated in nine equally sized quadrats/bog.

A before-and-after study in 2001–2002 in a historically mined bog in Quebec, Canada (4) reported that fen-characteristic plant species appeared following rewetting. These results are not based on tests of statistical significance. Before rewetting, no vegetation was present. Six months after rewetting, six local fen-characteristic plant species were present. Sixteen months after rewetting, five fen-characteristic species were present. Cover of fen-characteristic plants was 2% six months after rewetting and 10% sixteen months after. Note that the aim of this study was to create a fen, as the post-mining peat chemistry was more like a fen than a bog. In April 2001, a bog was rewetted by blocking the main drainage ditch, unblocking a water supply ditch and building embankments downslope of the bog to retain water. Afterwards, the water table was 1–65 cm below the peat surface during the growing season. In October 2001 and August 2002, cover of every plant species was estimated in ninety 30 x 30 cm quadrats (ten in each of nine 5 x 5 m plots). None of these plots were sown.

A replicated, paired, controlled study in 1997–2004 in a degraded raised bog in the Czech Republic (5) found that rewetted plots developed different plant communities, but with similar species richness, to plots that remained drained. Over four years after rewetting, the overall plant community composition significantly differed between rewetted and drained plots (data reported as graphical analyses). Plant species with greater cover in rewetted than drained plots included *Sphagnum* mosses, white sedge *Carex canescens*, sheathed cottongrass *Eriophorum vaginatum* and marsh thistle *Cirsium palustre*. Forest-characteristic species had lower cover in rewetted than drained plots. Rewetting had no effect on plant species richness, which fluctuated similarly over time in rewetted plots (5–8 species/m^2) and drained plots (4–6 species/m^2). In 1997 and 2000, three drainage ditches were blocked with a total of 17 dams, rewetting the peat above. In rewetted plots, the water table was 7 cm below the peat surface on average (vs drained plots: 15 cm below). Over the following 1–4 years, cover of every plant species was visually estimated in 17 pairs of 1 m^2 quadrats (one quadrat above and below each dam).

A before-and-after study in 2001–2004 in a degraded fen in Hungary (6) reported changes in cover of plant community types following rewetting. For example, in the driest areas (winter water level <30 cm) *Carex* sedge communities dominated 2% of quadrats before rewetting but 31% three years after. *Phalaris* grass communities dominated 60% of quadrats before rewetting but 2% three years after. In wetter areas (winter water level >30 cm), cover of aquatic vegetation was 0–8% before rewetting and 23–52% after, but for sedge communities was 31–50% before rewetting and 0–9% after. These results were not tested for statistical significance. In 2001, a drained fen was rewetted (water table raised to 0–116 cm above the peat surface) by building dykes to divert river water into it. Before rewetting in August

2001 and annually until 2004, the dominant plant community type was recorded in fourteen 25 m^2 quadrats along each of fourteen 100 m transects.

A before-and-after study in 2002–2004 in a degraded fen in California, USA (7) found that following rewetting, cover of peatland-characteristic sedges increased whilst cover of grass species preferring drier conditions decreased. Cover of three sedge species characteristic of wet peatlands increased (two significantly or marginally so), from 12–15% before rewetting to 13–20% one year after. Cover of three grass species that prefer drier conditions decreased (two significantly), from 2–6% before rewetting to 1–5% one year after. In July 2003, a fen was rewetted by blocking the main drainage ditch with metal dams. At the same time, channels were dug across a road that previously blocked surface water inflow. The water table was raised, ranging from 55 cm below the peat surface to 15 cm above during the summer. Vegetation cover was estimated in July before (2002) and after (2004) rewetting, in fifty-nine 10 m^2 plots.

A controlled, before-and-after study in 2000–2002 in a degraded fen in New Zealand (8) reported that following rewetting, total vegetation and upland plant cover decreased whilst wetland plant cover was stable. These results were not tested for statistical significance. Total vegetation cover declined in all four rewetted plots (before rewetting: 95–100%; one year after: 45–95%) but was stable in drained control plots (before: 92–100%; after: 90–100%). For two abundant, non-native, upland species, cover declined in all four rewetted plots (before: 5–40%; after: 0–5%) but was relatively stable in drained plots (before: 5–43%; after: 5–40%). For two abundant native species that only grow in wetlands, cover was stable in all rewetted plots (before: 22–90%; after: 24–90%) and all but one drained plot (where cover dropped from 13 to 0%). In March 2001, four plots within a fen were rewetted by blocking the main fen drain with soil dams. Nine plots remained drained. In 2000 and 2002, vegetation cover was estimated in four 4 m^2 quadrats/plot.

A replicated site comparison study in 2002 across five fen meadows in the Netherlands (9) found that rewetting had scale-dependent effects on plant species richness and diversity, and mixed effects on fen-characteristic species. At the site scale, rewetted meadows contained fewer plant species after four years than meadows that remained drained (25 vs 30 species/meadow). However, at the quadrat scale, rewetted meadows had significantly higher species richness than drained meadows (9 vs 7 species/m^2) and significantly higher diversity (data reported as a diversity index). Other reported data (not statistically tested) included abundance of sedges *Carex* spp. (rewetted: in 3–18% of quadrats; drained: in 10% of quadrats), abundance of common reed *Phragmites australis* (rewetted: 3%; drained: 0%) and number of fen-characteristic species (rewetted: 11/meadow; drained: 15/meadow). In 1998, four fen meadows were rewetted by isolating them from their drainage systems. A reference meadow remained drained. In spring 2002, vegetation cover was visually estimated in ten 1 m^2 quadrats/meadow. The water table was, on average, 20–30 cm below the peat surface in the rewetted meadows and 45 cm below the surface in the drained meadow.

A before-and-after study in 1995–2006 in a degraded rich fen in Sweden (10) reported that following rewetting, plant species richness, *Sphagnum* moss cover and tree cover increased, but cover of shrubs and purple moor grass *Molinia caerulea* decreased. These results were not tested for statistical significance. Plots contained 13–15 plant species before rewetting but 18–27 species four years after. In the plot where it occurred, cover of spiky bog moss *Sphagnum squarrosum* was 1% before rewetting but 13% after. Tree cover was 9–18% before rewetting but 13–20% after. In contrast, cover of purple moor grass was 58–74% before rewetting but only 32–62% after, and cover of shrubs 11–26% before rewetting but only 1–5% after. There was no clear change in cover of sedges *Carex* spp. (before: 0–1%; after: 0–2%) or two fen-characteristic moss species (0% before and after). In December 2002, the water table of a drained fen was raised approximately 17 cm by blocking a drainage ditch. Cover of every plant species was estimated in summer before (1995, 1997 or 2002) and after (2006) rewetting, in nine 1 m^2 quadrats in each of two plots.

A replicated, paired, controlled study in two degraded blanket bogs in Scotland, UK (11a) reported that rewetted plots developed greater cover of sheathed cottongrass *Eriophorum vaginatum*, and typically less cover of forest mosses, than drained plots. These results were not tested for statistical significance. In three of three comparisons, rewetted plots had greater cottongrass cover than drained plots after five years (rewetted: 19–45%; drained: 11–34%), but less plait moss *Hypnum cupressiforme* cover (rewetted: 18–44%; drained: 35–57%). Rewetting reduced cover of silk moss *Plagiothecum undulatum* in one of three comparisons, when plots remained forested (rewetted: 3%; drained: 6%) but had no additional effect in plots where trees were felled or removed (rewetted: 0.6–0.8%; drained: 0.5–0.6%). Six blocks of six 40 x 100 m plots were established in drained bogs forested with spruce and pine. Between 1996 and 1998, six treatments were replicated once/block: rewetting, rewetting with tree felling, rewetting with tree removal, tree felling only, tree removal only, no intervention. Rewetting was achieved by damming plough furrows every 20 m. In rewetted plots, the water table was 8–32 cm below the peat surface during the growing season (vs drained plots: 11–38 cm below). Vegetation cover was recorded five years after intervention (details not reported).

A replicated, paired, controlled study in a degraded raised bog in Scotland, UK (11b) reported that blocking plough furrows to help rewet the bog had no (or no consistent) effect on vegetation cover in plots where trees had been felled. These results were not tested for statistical significance. After five years, plots with blocked and open furrows had similar cover of heather *Calluna vulgaris* (3–15% vs 3–14%) and sheathed cottongrass *Eriophorum angustifolium* (20–48% vs 15–45%). Grass cover was similar in blocked and open plots when trees had been removed (3 vs 2%), but higher in blocked plots when all tree debris was left in place (6 vs 1%) and lower in blocked plots when tree tops were left in place (7 vs 10%). Grass was mostly wavy hair grass *Deschampsia flexuosa*. Twelve pairs of 18 x 20 m plots were established in a drained, pine-forested bog. Between 1996 and 1998, plough furrows were blocked in one plot/pair but left open in the other. The water table depth was similar under both treatments (0–22 cm below the peat surface). Trees were felled in all plots, with debris left in place (four pairs), tree tops left in place (four pairs) or all debris removed (four pairs). Vegetation cover was recorded five years after intervention (details not reported).

A replicated, paired, controlled, before-and-after study in 2002–2005 in two degraded rich fens in Sweden (12) reported that rewetting alone led to small changes in plant community composition and cover. These results are not based on tests of statistical significance. The overall composition of the plant community changed over three years following rewetting (data reported as a graphical analysis). Cover of sedges *Carex* spp. increased in rewetted plots (from 0–2% before rewetting to 1–8% three years after) but was stable in drained plots (0–1%). Purple moor grass *Molinia caerulea* was common in one fen, where cover decreased in rewetted plots (from 50 to

30%) but was stable in drained plots (50–55%). *Sphagnum* mosses were common in the other fen, where cover increased in rewetted plots (from 14 to 25%) but decreased in drained plots (from 43 to 28%) – although responses differed between species. In spring 2003, one 50 x 150 m^2 plot in each fen was rewetted by blocking a drainage ditch (water table raised approximately 10 cm). An adjacent plot in each fen remained drained. Both plots were also cleared of trees. Vegetation cover was estimated, in 0.25 m^2 quadrats, in the central 100 m^2 of each plot: sixteen quadrats in 2002 across the whole 100 m^2, and four quadrats in 2005 within subplots that received no additional treatment. This study was based on the same experimental set-up as (17) and (21).

A site comparison study in 2004 in two peatlands in Finland (13) reported that a rewetted peatland developed a different plant community to a pristine peatland, with lower plant species richness, lower shrub cover and greater sedge/cottongrass cover (but similar forb and moss cover). Most of these results were not tested for statistical significance. After 10 years, the overall plant community composition differed between the rewetted and pristine peatland (data reported as a graphical analysis and similarity index). The rewetted peatland contained only 15 plant species (vs 18 in the pristine peatland) and 5 species/60 x 60 cm quadrat (vs 9 species). In the rewetted peatland there were no dwarf shrubs (vs 3% cover in the pristine peatland) but sedge/cottongrass cover was 20% (vs 4%). Both peatlands had similar forb cover (5 vs 5%) and *Sphagnum* moss cover (84 vs 90%), and there was no significant difference in total moss cover (89 vs 90%). In 2004, cover of every plant species was recorded in 15 quadrats, each approximately 60 x 60 cm. Nine quadrats were in a historically mined peatland, rewetted in 1994 (water table 9 cm above peat surface during summer). Six quadrats were in a nearby pristine peatland with similar physical conditions (but a lower water table: 6 cm below surface). This study used the same rewetted peatland as (2).

A before-and-after study in 1996–2007 in a historically mined raised bog in Germany (14) found that following rewetting, there were increases in the number of plant species, moss cover and sedge/rush cover, but decreases in shrub and tree cover. The number of plant species on the bog increased from 157 before rewetting to 208 ten years after. Over the same time period, total moss cover increased from 31 to 44% and *Sphagnum* moss cover increased from 15 to 25%. Total herb cover did not change significantly over time (67% before and after). However, sedge/rush cover increased from 7–10% to 19–27% (not statistically tested). Shrub cover decreased from 41 to 22%. Tree cover decreased from 31 to 8%. In 1997, a bog was rewetted by blocking drainage ditches in and around it (water table raised to 5–55 cm below the peat surface). Plant species were counted and vegetation cover estimated before (1996) and after (2007) ditch blocking, in 35 permanent 25 m^2 plots. This study was based on the same experimental set-up as (16).

A replicated site comparison study in 2007 in four blanket bogs in Scotland, UK (15) reported that two rewetted bogs had, after 3–11 years, similar cover of bog-characteristic vegetation and open water (6–26%) to bogs that remained drained (5–16%). This result is not based on a test of statistical significance. Additionally, in one bog (Cross Lochs), transects rewetted eleven years before measurement had greater cover of bog plants/open water (26%) than transects rewetted only four years before (10%). Between 1996 and 2004, two drained bogs were rewetted by blocking most of their drainage ditches with peat and plastic dams. Two other bogs remained drained (ditches were not blocked). In summer 2007, cover of vegetation and open water were recorded in each bog along 30–60 randomly placed 1 m transects. Cover of bog characteristic *Sphagnum* moss species, common cottongrass *Eriophorum angustifolium*, dead heather *Calluna vulgaris* and open water were combined into a 'bog recovery index' for analysis.

A site comparison study in 2007 in two bogs in Germany (16) found that a fully rewetted bog contained more plant species, greater moss cover and greater cover of some herbs than a partially rewetted bog, but less tree and rush cover. After 10 years, there were more plant species on the fully rewetted bog (208) than the partially rewetted bog (68). The fully rewetted bog also had greater cover of mosses overall (44 vs 4%) and *Sphagnum* mosses (25 vs 14%). Beaked sedge *Carex rostrata*, purple moor grass *Molinia caerulea* and one of two *Eriophorum* cottongrass species were more abundant, relative to other plant species, in the fully rewetted bog than the partially rewetted bog (reported as an abundance index). The fully rewetted bog had less cover of rushes (19 vs 39%) and trees (22 vs 57%). Both bogs had similar cover of herbs overall (67 vs 65%) and shrubs (8 vs 6%). In 1997, drainage ditches in and around both bogs were blocked. In one bog all blockages were successful (water table 17–25 cm below peat surface, on average) but in the other bog only some blockages were successful, so the water table was lower (40–50 cm below surface). In 2007, plant species were counted and vegetation cover estimated in 25 m^2 plots: 35 on the fully rewetted and 21 on the partially rewetted bog.

A replicated, paired, controlled, before-and-after study in 2002–2010 in three degraded rich fens in Sweden (17) found that following rewetting bryophyte and sedge cover increased, but there was no change in species richness, fen-characteristic plant cover or grass cover. Cover of wetland-characteristic bryophytes increased from 33% before rewetting to 46% eight years after. *Sphagnum* moss cover increased from 10 to 18%. Sedge cover increased from 1 to 3%. There was no significant change in cover of fen-characteristic plants or grasses (data not reported) or plant species richness (from 8 to 10 species/0.25 m^2). In plots that remained drained, none of the metrics changed significantly over the eight years. In winter 2002/2003, one 100 x 150 m plot in each drained fen was rewetted by blocking a drainage ditch (water table raised by 12–25 cm). An adjacent plot in each fen remained drained. Trees were also removed from half of each plot. Between 2002 (before intervention) and 2010, cover of every plant species was estimated at 40 points/plot, in 0.25 m^2 quadrats. This study was based on the same experimental set-up as (12) and (21).

A before-and-after study in degraded peatland in China (18a) reported that following rewetting, cover of wetland-characteristic plants developed. No statistical tests were carried out. Before rewetting, wetland-characteristic plants were confined to drainage ditches (precise cover not reported). After rewetting, wetland-characteristic plants were observed across the peatland. Dominant plants in each part of the peatland were spikesedge *Heleocharis valleculosa* (80% cover), sedge *Carex muliensis* (60–70% cover) and Kneiff's feather moss *Leptodictyum riparium* (15–80% cover). The blocked drainage ditches were dominated by floating bur-reed *Sparganium angustifolium* (30% cover). Within the drained and grazed Riganqiao peatland, two drainage ditches were blocked with 12 wooden dams. The water table rose above the peat surface in most areas. After rewetting, vegetation cover was visually estimated in five areas of the peatland (precise methods and dates not reported).

A study in a degraded peatland in China (18b) reported that following rewetting, the peatland was colonized by wetland-characteristic plants. No statistical tests were carried out. Wetland-characteristic plants were observed across the rewetted peatland. Dominant plants in different parts of the peatland included sedges *Kobresia capillifolia* (10–60% cover) and *Carex pamirensis* (50% cover), rush *Blysmus sinocompressus* (20% cover) and marsh arrowgrass *Triglochin palustre* (15% cover). In 2004, the main drainage ditch in Dazhasi peatland was blocked with 19 sandbag dams, raising the water table. The dams failed over winter but were rebuilt each spring. After rewetting (year not reported), vegetation cover was visually estimated in five 4–100 m^2 areas of the peatland.

A replicated site comparison study in 2009 in 11 rich fens in Belgium, Poland and the Netherlands (19) found that rewetted fens had similar total plant species richness to fens that had never been drained, but lower fen-characteristic species richness, greater herb cover and lower moss cover. Rewetted fens contained a similar total number of plant species to never-drained fens (27 vs 30 species/25 m^2) but fewer fen-characteristic species (7 vs 15 species/25 m^2). Rewetted fens had greater overall herb cover (52 vs 28%) and tall sedge/rush cover (41 vs 18%), but less cover of mosses overall (50 vs 90%) and fen-characteristic mosses (4 vs 40%). Cover of fen-characteristic vascular plants was similar in rewetted and never-drained fens (34 vs 33%). In summer 2009, cover of every plant species was estimated in sixteen 25 m^2 plots: nine plots across five rewetted fens in Belgium and the Netherlands, and seven plots across six never-drained fens in Poland. Details of rewetting were not reported, but the water table in all fens was <10 cm below the peat surface. Plots experienced a range of mowing regimes. The rewetted fens contained more iron and phosphorous than the never-drained fens, which may have affected the vegetation.

A replicated, before-and-after, site comparison study in 2011–2013 in a degraded raised bog in Latvia (20) reported that rewetting had no effect on plant species richness, increased cover of sheathed cottongrass *Eriophorum vaginatum,* white beak sedge *Rhynchospora alba* and *Sphagnum* moss, and reduced cover of heather *Calluna vulgaris* (but not other trees/shrubs). Most of these results were not tested for statistical significance. Over six months, rewetting had no significant effect on plant species richness on three of four transects (before: 13.6–15.2 species/20 m^2; after: 13.6–15.8 species/20 m^2) but increased cover of sheathed cottongrass in 19 of 21 quadrats (by 1–5%). Over 18 months, rewetting increased cover of white beak sedge (before: 1%; after: 4%) and *Sphagnum* moss (before: 53%; after: 72%), but reduced heather cover (before: 84%; after: 68%) and had no effect on other tree/shrub cover (before: 20%; after: 22%). When vegetation cover changed, it became more like a pristine bog (with 80% *Sphagnum*, 18% sedge and 9% heather cover). In early 2012, drainage ditches were blocked in a bog remnant. In summer 2012 and 2013, cover of every plant species was estimated in the rewetted bog (in twenty-one 4 m^2 quadrats) and in a nearby undisturbed bog (details not reported).

A replicated, paired, controlled, before-and-after, site comparison study in 2002–2010 involving three degraded rich fens in Sweden (21) reported that rewetting increased plant species richness but found that it had no effect on vegetation height. After eight years, rewetted plots had higher plant species richness than drained plots (not tested for statistical significance). This effect was larger in plots that remained forested (rewetted: 13; drained: 9 species/0.25 m^2) than in plots previously cleared of trees (rewetted: 14; drained: 13 species/0.25 m^2). Rewetting had no effect on vegetation height: it was similar in both treatments after eight years (rewetted: 5–6 m; drained: 5–6 m) and did not change significantly over time (data not reported). For comparison, a nearby natural (undrained and unforested) fen contained 9 plant species/0.25 m^2 and had a canopy height of 1 m. These were significantly greater in the rewetted plots. In winter 2002/2003 at each of three sites, two adjacent 100 x 150 m plots were established: one rewetted above a ditch blockage, and one drained below. Trees were also removed from half of each plot. Before intervention in 2002, then until 2010, plant species and canopy height (ignoring trees present before intervention) were recorded at 40 points/plot, in 0.25 m^2 quadrats. The natural fen was sampled in 1978. This study was based on the same experimental set-up as (12) and (17).

A before-and-after, site comparison study in 2004–2009 in a degraded fen in Poland (22) found that in a rewetted area (also cleared of shrubs and mown), the plant community composition changed in favour of fen and wet meadow species. Over five years, the overall plant community composition in a managed area became more like target fen meadow vegetation (data reported as a graphical analysis; change not tested for statistical significance). The abundance of fen and wet meadow plant species, including sedges *Carex* spp., increased in the managed area but did not change in the target area (data reported as abundance indices). In 2004, 0.7 ha of drained, overgrown fen was rewetted by blocking its connection to a drainage ditch. After rewetting, the water table was 0–16 cm below the peat surface (summer average). The fen was also cleared of willow *Salix cinerea* shrubs, then mown annually. The study does not distinguish between the effects of these interventions. The managed area was compared to 0.9 ha of target, shrub-free, fen meadow vegetation (retained in depressions during the drained period, but also affected by the rewetting and mown every other year). Each year between 2004 (before intervention) and 2009, vegetation cover was estimated in 18–22 plots/area. Plots were 20 x 20 m.

A replicated before-and-after study in 2010–2013 in three degraded raised bogs in Latvia (23) reported that rewetting had no effect on vegetation cover after one year. These results were not tested for statistical significance. In all three monitored sites, vegetation cover was similar in the three years before rewetting and the year after rewetting. This was true for heather *Calluna vulgaris* (before: 51–61%; after: 48–60%), *Sphagnum* mosses (before: 18–30%; after: 19–28%) and three other moss species (3–16% before and after). In 2012, drainage ditches were blocked in three degraded bogs. Each year between 2010 and 2013, vegetation cover was visually estimated in 25–30 permanent quadrats/bog. Quadrats were circular (4 m diameter) and arranged along transects perpendicular to the blocked ditches.

A controlled study in 2006–2012 in a historically mined raised bog in Latvia (24) reported that following rewetting, cover of feathery bog moss *Sphagnum cuspidatum* and white beak sedge *Rhynchospora alba* increased whilst cover of heather *Calluna vulgaris* decreased. Most of these results were not tested for statistical significance. Cover of feathery bog moss was 2% one year after rewetting but 28% six years after. No other *Sphagnum* species colonized. Cover of white beak sedge was 2% one year after rewetting but 39% six years after. Over the same period, heather cover decreased significantly from 35 to 19%. However, heather cover also decreased significantly in plots that were not rewetted (data not reported). In 2006, drainage ditches on part of a historically milled bog were blocked, raising the water table by 60 cm. In the rest of the bog, drainage ditches were left unblocked. Each year between 2007 and 2012, vegetation cover was estimated in permanent quadrats (4 m diameter circles): twenty-one in the rewetted area and seven in the drained area.

A replicated before-and-after study in 2007–2008 in 23 degraded fens in Germany (25) reported changes in the cover of plant community types following rewetting. These results were not tested for statistical significance. Before rewetting, all sites were dry grassland (precise cover not reported). After rewetting, the fens remained dominated by grasses (most common vegetation type in all 23 fens; overall cover 48%) but cover of other plant groups had increased, including trees/shrubs (in 23 fens; overall cover 13%), common reed *Phragmites australis* (in 22 fens; overall cover 5%), cattail *Typha latifolia* (in 23 fens; overall cover 4%) and sedges *Carex* spp. (in 23 fens; overall cover 2%). Between 1995 and 2008, the 23 drained fens were rewetted by stopping their pump drainage systems (water table raised to 20–50 cm above peat surface). In 2007 and 2008, cover of vegetation groups was recorded in field surveys and/or from satellite images.

A replicated before-and-after study in 2006–2013 seven degraded peatlands in England, UK (26) reported that following rewetting, plant community types and purple moor grass *Molinia caerulea* abundance typically did not change, but *Sphagnum* mosses became more abundant. These results were not tested for statistical significance. Initially, all seven sites contained wet heath plant communities. After 2–6 years, four rewetted sites were still wet heaths but three had developed peatland plant communities. Purple moor grass abundance was similar (present in 95–100% of quadrats) before and after rewetting in all sites. *Sphagnum* moss species became more abundant after rewetting in most (21 of 34) comparisons. Abundance of blunt-leaved bog moss *Sphagnum palustre* consistently increased (six of six comparisons). Sedge *Carex* spp. and common cottongrass *Eriophorum angustifolium* abundance showed mixed responses depending on species and site. Between 2008 and 2013, one drainage ditch was blocked in each of seven peatlands. Vegetation (species presence/absence) was recorded before ditch blocking and 3–7 years after. In each site 120–160 quadrats (0.25 m^2), arranged along a 30–40 m transect perpendicular to the blocked ditch, were surveyed in summer or autumn.

A replicated, paired, site comparison study in 1993–2010 in three historically mined bogs in Quebec, Canada (27) found that rewetted areas typically had similar moss, herb and tree cover to areas that remained drained, but less shrub cover. In most cases, there was no difference between areas rewetted for 4–17 years and areas that remained drained, for *Sphagnum* moss cover (17 of 24 comparisons; rewetted: 0–21%; drained: 0–26%), other moss cover (8 of 15 comparisons; rewetted: 0–19%; drained: 0–31%), herb cover (8 of 12 comparisons; rewetted: 0–42%; drained: 0–24%) and tree cover (11 of 12 comparisons; rewetted: 0–8%; drained: 1–17%). However, in most cases shrub cover was lower in rewetted areas (11 of 21 comparisons; rewetted: 0–49%; drained: 1–71%). In the remaining comparisons, rewetted plots had greater *Sphagnum* and herb cover, lower tree cover, greater or lower moss cover, and similar shrub cover. Between 1993 and 2006, parts of three bogs were rewetted by blocking drainage ditches (water table 7–44 cm above peat surface, on average, during summer). Each bog also contained areas that remained drained (ditches not blocked; water table 2–8 cm above surface). In 2010, in each rewetted and drained area, all plant species touching 400–1,800 evenly spaced points were recorded.

A replicated site comparison study in 2009 in 36 forested fens in Finland (28) found that rewetting changed the plant community composition towards a more natural state, but had no effect on plant richness or diversity, tree volume or vegetation cover. After 1–14 years, the overall plant community composition in rewetted sites was intermediate between, but significantly different from, both drained and natural sites (data reported as a graphical analysis). In contrast, rewetting had no significant effect on plant species richness (rewetted: 31; drained: 30 species/site), plant diversity (reported as a diversity index), tree volume (rewetted: 235; drained: 335 m^3/ha) and *Sphagnum* moss cover (rewetted: 25%: drained: 9%). Also similar between sites, but not statistically tested, were other moss cover (rewetted: 22%: drained: 25%), shrub cover (rewetted: 9%: drained: 8%) and herb cover (rewetted: 5%: drained: 2%). Compared to natural sites, rewetted sites had lower *Sphagnum* moss cover (natural: 46%) but greater cover of other mosses (natural: 3%) and greater plant diversity. Of the 36 forested fens studied, 18 had been rewetted in 1995–2008 by filling or blocking drainage ditches (water table raised to 15 cm below the peat surface). Nine fens remained drained (ditches open; water table 40 cm below surface). Nine fens had never been drained (water table 17 cm below surface). In 2009, vegetation cover and species were recorded in 72 circular (30 cm diameter) quadrats/site. Tree volume was measured in one 30 x 30 m plot/site. This study used the same sites as (30).

A replicated site comparison study in 1998–2012 in eight fens in Germany (29) found that rewetted fens contained a similar plant community of similar height to both drained and natural fens, and had similar plant species richness to drained (but not natural) fens. After 14 years, the overall plant community composition in rewetted fens was not significantly different from drained or near-natural fens (but intermediate between the two; data reported as similarity indices). Vegetation height did not differ significantly between fen types (rewetted: 118 cm; drained: 56 cm; near-natural: 128 cm). Plant species richness was similar in rewetted and drained fens (both 18 species/4 m^2), but lower than in near-natural fens (32 species/4 m^2). Eight neighbouring fens were compared: two rewetted in 1998 (water table 38 cm below peat surface during summer), three that remained drained (water table 78 cm below surface) and three near-natural (never substantially drained; water table 13 cm below surface). In August and September 2012, cover of every plant species was estimated in thirty 4 m^2 quadrats/fen type. Overall vegetation height was measured at 150 points/fen type.

A replicated site comparison study in 2011–2012 in 36 forested fens in Finland (30) found that in rewetted sites, *Sphagnum* moss growth was greater than in drained sites, and similar to undrained sites. After 3–16 years, *Sphagnum* biomass growth in rewetted sites (147 g/m^2/year) was significantly greater than in sites that remained drained (76 g/m^2/year) and not significantly different to never-drained sites (128 g/m^2/year). The same was true for length growth (rewetted: 6; drained: 3; never-drained: 5 g/m^2/year). Of the 36 forested fens studied, 18 had been rewetted in 1995–2008 by filling or blocking drainage ditches, nine remained drained (ditches open) and nine had never been drained. In May 2011, nine 13 x 13 cm plastic nets were installed on the rewetted peat in each site. In May 2012, all *Sphagnum* growing above each net was harvested, then dried and weighed. Stem length was measured for 20 shoots/net. This study used the same sites as (28).

A before-and-after study in 2008–2013 in a degraded poor fen in Spain (31) reported that following rewetting (along with cattle exclusion), cover of rushes *Juncus* spp. increased and new populations of *Sphagnum* moss appeared. No statistical tests were carried out. Before intervention, the fen was covered by dryland grasses and forbs, with no *Sphagnum*. Four years after intervention, 81% of the fen area was dominated by rushes: common rush *Juncus effusus* with some sharp-flowered rush *Juncus acutiflorus*. *Sphagnum* mosses also appeared in 3 of 10 monitored quadrats. In 2009, a drained fen was rewetted by blocking drainage ditches, removing a drainage pipe and building a new inflow ditch. The fen was also fenced to exclude cattle. The study does not distinguish between the effects of rewetting and cattle exclusion. Vegetation cover was estimated in 2008 (before restoration) and 2013, in ten permanent quadrats (size not reported) and from aerial photographs.

A replicated before-and-after study in 2007–2015 seven degraded peatlands in England, UK (32) reported that following rewetting, plant species richness consistently increased but the plant community type did not change. These results were not tested for statistical significance. Plant species richness increased in all seven sites, from 11–39 species/site before rewetting to 17–49 species/site after 2–7 years. In contrast, the plant community type did not change in any site. Both before and after rewetting, four sites contained wet heath communities, two sites contained dry heath communities and one site contained a dry grassland community. Between 2008 and 2013, one drainage ditch was blocked in each site. Vegetation was recorded before ditch blocking (as presence/absence of species) and 2–7 years after (as cover of every species). In each site 120–160 quadrats (0.25 m^2), arranged along a 30–40 m transect perpendicular to the blocked ditch, were surveyed in summer or autumn.

A replicated before-and-after study in 1999–2014 in two historically mined bogs in Sweden (33) reported that rewetted bogs developed plant communities that included some key bog species. Before rewetting, both bogs were bare peat. In Västkärr bog, vegetation developed within one year after rewetting. The overall plant community composition changed significantly over the 14 measured years (data reported as a graphical analysis). During this period, there were 2–6 plant species/m^2 and vegetation cover was 30–112%. After 14 years, vegetation cover included sedges *Carex* spp. (23%), duckweed *Lemna minor* (15%), cattail *Typha latifolia* (10%) and common reed *Phragmites australis* (1%). In Porla bog, vegetation cover developed 7–14 years after rewetting. During this period, the overall plant community composition did not change significantly (data reported as a graphical analysis). There were 2–4 plant species/m^2 and vegetation cover was 40–77%. After 14 years, vegetation cover included *Eriophorum* spp. (13–32%), *Sphagnum* mosses (20%), other mosses (<1%), sedges (2%) and common reed (2%). In 1999, both drained bogs were cleared of existing vegetation then rewetted (Västkärr by filling ditches and ceasing pumping, Porla by restoring inflow). Between 2000 and 2014, cover of every plant species was estimated in 1 m^2 quadrats: 1–32 quadrats/year/bog. The water table in the sampled areas was 0–20 cm above the peat surface.

A site comparison study in 2008–2014 in a historically mined bog in Quebec, Canada (34) reported that a rewetted area developed a different plant community to, with less vegetation cover than, nearby natural fens. These results were not tested for statistical significance. Note that the aim of this study was to create a fen, as the post-mining peat chemistry was more like a fen than a bog. After five years, the rewetted area contained a different overall plant community to three nearby natural fens (data reported as a graphical analysis). In the rewetted area, *Sphagnum* moss was absent (vs 15–25% in natural fens), other moss cover only 8% (vs 12–55%) and vascular plant cover only 24% (vs 59–86%). The rewetted area was dominated by woolgrass *Scirpus cyperinus* (19% cover; natural fens: 0%) and bog myrtle *Myrica gale* (8% cover; natural fens: 4–19%). In winter 2009/2010, part of a historically mined bog (abandoned for nine years) was rewetted by blocking drainage ditches with peat. Vegetation cover was estimated in 2008 (donor fen: in 16 quadrats along a transect) or 2014 (rewetted area: in six 25 m^2 plots).

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8.2 Irrigate peatland

Ⓑ Ⓕ Ⓢ

- **Two studies** examined the effect of irrigation (without planting) on peatland vegetation. One study was in a bog[1] and one was in a fen[2].
- **Vegetation cover (2 studies):** One replicated, paired, controlled, before-and-after study in a bog in Canada[1] found that irrigation increased the number of *Sphagnum* moss shoots present after one growing season, but had no effect after two. One before-and-after study in Germany[2] reported that an irrigated fen was colonized by wetland- and fen-characteristic herbs, whilst cover of dryland grasses decreased.

Background

To avoid desiccation of vegetation added to the surface of peatlands, irrigation systems such as sprinklers could be used (Rochefort & Bastien 1998). Water could be recirculated from drainage ditches or ponds on the peatland. Irrigation would maintain a damp peat surface. Irrigation can be expensive so may be best used as a short-term intervention to kick-start restoration.

CAUTION: A suitable water source, with the right level of nutrients and acidity/alkalinity, must be chosen to avoid altering chemical conditions on the peatland (Lamers *et al.* 2002). For example, bogs should only be irrigated with water stored on the bog, not ground water. Taking water for irrigation might reduce water levels in neighbouring wetlands.

Related interventions: rewet peatlands by raising the water table rather than irrigating the surface (Section 8.1); irrigation to complement planting (Section 13.8).

Lamers L.P., Smolders A.J.P. & Roelofs J.G.M. (2002) The restoration of fens in the Netherlands. *Hydrobiologia*, 478, 107–130.

Rochefort L. & Bastien D.F. (1998) Réintroduction de sphaignes dans une tourbière exploitée: évaluation de divers moyens de protection contre la dessiccation (Reintroduction of *Sphagnum* to an exploited bog: evaluation of various methods for protection against desiccation; in French). *Écoscience*, 5, 117–127.

A replicated, paired, controlled, before-and-after study in 1993–1994 in a historically mined bog in Quebec, Canada (1) found that irrigated plots contained more *Sphagnum* moss shoots than unirrigated plots after one growing season, but a similar number of *Sphagnum* shoots after two. After one growing season there were more *Sphagnum* shoots in irrigated plots (210 shoots/m^2) than plots that were not irrigated (75 shoots/m^2). However, after two growing seasons the number of moss shoots did not significantly differ between treatments (irrigated: 80; not irrigated: 50 shoots/m^2). In spring 1993, three pairs of plots were established on slightly drained, bare peat. Three plots (one plot/pair) were irrigated during the summer, using sprinklers and water stored on the bog. The other plots were not irrigated. In autumn 1993 and 1994, all *Sphagnum* shoots were counted in forty 30 x 30 cm quadrats/plot.

A before-and-after study in 1996–1998 in a degraded fen in Germany (2) reported that irrigated plots developed cover of wetland- and fen-characteristic herbs at the expense of dry grassland species. All data were reported as graphical analyses and the results were not tested for statistical significance. Over two years of irrigation, cover of fen-characteristic forbs increased. The same was true for cover of wetland-characteristic species like cattail *Typha latifolia* and common rush *Juncus effusus*. Meanwhile, cover of dry grassland species such as tall fescue *Festuca arundinacea* decreased. No colonisation by sedges *Carex* spp. or common reed *Phragmites australis* was observed. Within the irrigated fen, plant communities differed between drier areas (high water table but never flooded) and wetter areas (sometimes flooded). In 1996, the surface of a drained fen was irrigated with lake water. Vegetation cover was recorded before irrigation (1996) and after one or two years or irrigation (1997, 1998) in four representative 16 m^2 plots.

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(2) Richert M., Dietrich O., Koppsich D. & Roth S. (2000) The influence of rewetting on vegetation development and decomposition in a degraded fen. *Restoration Ecology*, 8, 186–195.

8.3 Reduce water level of flooded peatlands

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of reducing the water level in unnaturally flooded peatlands.

Background

Pools may naturally form on the surface of peatlands. However, peatland vegetation can be destroyed by deep, long-term flooding e.g. above dams constructed for electricity generation (Kelly *et al.* 1997) or following extreme weather events. Lowering the water level could allow the former peatland vegetation to recover.

Kelly C.A., Rudd J.W.M., Bodalay R.A., Roulet N.P., St. Louis V.L., Heyes A., Moore T.R., Schiff S., Aravena R., Scott K.J., Dyck B., Harris R., Warner B. & Edwards G. (1997) Increases in fluxes of greenhouse gases and methyl mercury following flooding of an experimental reservoir. *Environmental Science and Technology*, 31, 1334–1344.

8.4 Restore natural water level fluctuations

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of restoring natural water level fluctuations *per se*.

Background

The water table in peatlands may naturally move up and down depending on the season, rainfall events and regional ground-water levels. Management to maintain stable water levels, for example in fens used for agriculture, can lead to nutrient enrichment and reduced plant diversity. Nutrient enrichment can occur because of chemical reactions in dry peat (especially the release of phosphorous), or because input water used to control the water level contains nutrients (Lamers *et al.* 2002). Reinstating water level fluctuations could allow natural communities to recover.

Related intervention: rewetting, which may restore a naturally fluctuating water table without examining the effect of these fluctuations *per se* (Section 8.1).

Lamers L.P.M., Smolders A.J.P. & Roelofs J.G.M. (2002) The restoration of fens in the Netherlands. *Hydrobiologia*, 478, 107–130.

Interventions: Modified vegetation management

8.5 Cut/mow herbaceous plants to maintain or restore disturbance

B Ⓕ S

- **Fourteen studies** examined the effect on peatland vegetation of cutting/mowing to maintain or restore disturbance. All 14 studies were in fens or fen meadows. Most studies examined the effect of annual mowing regimes, but three[11,13,14] examined the effect of single cuts. *N.B. Section 9.4 considers cutting/mowing in peatlands with no clear history of disturbance.*
- **Plant community composition (7 studies):** Six replicated studies in fens and fen meadows in the UK[1], Belgium[2,3], Germany[5,6] and the Czech Republic[8] reported that mowing altered the overall plant community composition (compared to no mowing[2,5,8], before mowing[1,3,8] or grazing[6]). One site comparison study in Poland[10] reported that mowing a degraded fen (along with other interventions) made the plant community more similar to a target fen meadow.
- **Characteristic plants (5 studies):** Four studies (including one replicated, paired, controlled, before-and-after) in fens and fen meadows in Switzerland[4], Germany[5], the Czech Republic[8] and Poland[10] found that cutting/mowing increased cover of fen- or wet meadow-characteristic plants. One replicated before-and-after study in fens in the UK[13] found that a single mow typically had no effect on cover of fen-characteristic plants. In Poland[10] and the UK[13], the effect of mowing was not separated from the effects of other interventions.
- **Moss cover (6 studies):** Of six studies (five replicated, paired, controlled) in fens or fen meadows, four in Belgium[2], Switzerland[4,7] and the Czech Republic[8] found that mowing increased

total moss or bryophyte cover. Two studies in Poland[11] and the UK[14] found that a single mow typically had no effect on bryophyte cover (total[14] or hollow-adapted mosses[11]).

- **Herb cover (8 studies):** Six replicated studies (three also randomized and controlled) in fens and fen meadows in Belgium[2,3], Germany[5], Poland[11] and the UK[13,14] found that mowing reduced cover or abundance of at least one group of herbs (including bindweed[2,3], reeds[5], sedges[5,11,14], purple moor grass[13] and grass-like plants overall[13,14]). One before-and-after study in a fen in Poland[10] found that mowing (along with other interventions) increased sedge cover. One replicated, randomized, paired, controlled study in fen meadows in Switzerland[7] found that mowing had no effect on overall herb cover.
- **Tree/shrub cover (3 studies):** Of three replicated studies in fens, two in the UK[13,14] found that a single mow (sometimes[13] along with other interventions) reduced shrub cover. However, one study in Poland[11] found that a single mow had no effect on shrub cover.
- **Vegetation structure (7 studies):** In the following studies, vegetation structure was measured 6–12 months after the most recent cut/mow. Three replicated studies in fens in Poland[11] and the UK[13,14] reported that a single mow (sometimes[13] along with other interventions) had no, or no consistent, effect on vegetation height. One replicated, paired, site comparison study in fen meadows in Switzerland[4] found that mowing reduced vegetation height. Three studies (including two replicated, paired, site comparisons) in fen meadows in Switzerland[4], Poland[9] and Italy[12] found mixed effects of mowing on vegetation biomass (total[4,9], sedge/rush[4], moss[9] or common reed[12]). One replicated, paired, site comparison study in Germany[6] reported that mown fen meadows had similar vegetation structure to grazed meadows.
- **Overall plant richness/diversity (11 studies):** Eight studies in fens and fen meadows in the UK[1], Belgium[2,3], Switzerland[4,7], Germany[6], the Czech Republic[8] and Poland[9] found that mowing/cutting increased plant species richness (compared to no mowing[2,4,7,8,9], before mowing[1,3,8] or grazing[6]). Three studies (including two replicated, randomized, paired, controlled) in fens in Poland[11] and the UK[13,14] found that a single mow (sometimes[13] along with other interventions) typically had no effect on plant species richness and/or diversity.

Background

Regular disturbance may maintain vegetation in a desirable, semi-natural state – particularly in fen meadows and some fens (Middleton 2012). Disturbance can clear dominant plants, maintain light availability and control nutrient levels. This can favour a plant community rich in plant species and/or peatland characteristic species. Therefore, conservationists may sometimes want to actively maintain or restore disturbance. Mowing or cutting (including machine mowing, hand clipping, strimming and scything) might be one way to do this.

Cutting itself may be the desirable, traditional disturbance. Some fens and fen meadows have been mown for hundreds of years, to produce animal feed or bedding (Güsewell 2003). However, many historically cut peatlands have been abandoned over the past 60 years, especially those in remote areas (Middleton *et al.* 2006).

CAUTION: Disturbance is not desirable on all peatlands. Fen meadows and some fens may benefit from disturbance. Many other fens, bogs and peat swamps will not. Where cutting is desirable, heavy machinery could damage the peatland surface and vegetation: cutting by hand or with specialized vehicles might cause less damage.

Related interventions: interventions to reduce the impacts of vehicles used for mowing (Sections 6.3 and 6.4); cutting large trees/shrubs, to maintain or restore disturbance (Section 8.7); change season of cutting/mowing (Section 9.5).

Güsewell, S. (2003) Management of *Phragmites australis* in Swiss fen meadows by mowing in early summer. *Wetlands Ecology and Management*, 11, 433–445.

Middleton B.A. (2012) Rediscovering traditional vegetation management in preserves: trading experiences between cultures and continents. *Biological Conservation*, 158, 750–760.

Middleton B., Holsten B. & van Diggelen R. (2006) Biodiversity management of fens and fen meadows by grazing, cutting and burning. *Applied Vegetation Science*, 9, 307–316.

A replicated before-and after study in 1980–1982 in a fen meadow in England, UK (1) reported that after reinstating annual summer mowing, the plant community composition changed and species richness increased. These results are not based on tests of statistical significance. The overall composition of the plant community changed over two years of annual mowing (data reported as a graphical analysis). There were 1.1–2.3 plant species/250 cm^2 before mowing, but 1.5–3.6 species/250 cm^2 after two years of annual mowing. Amongst mown plots, species richness increased more in July-mown subplots than May-mown subplots, but community composition changes were similar (see Section 9.5). In 1980–1982, traditional annual summer mowing was reinstated in 10 plots across two fen meadow community types. In each plot, one random 25 m^2 subplot was mown in May, one mown in July and one mown in May and July. Cuttings were removed. Immediately before each mowing, vascular plant species were recorded in sixteen 250 cm^2 quadrats/subplot. Prior to the study, the meadow had been mown every 3 years (rather than every year).

A replicated, paired, controlled study in 1977–1982 in a degraded fen in Belgium (2) reported that resuming winter mowing changed plant community composition, increased species richness and bryophyte cover, and reduced cover of one of three dominant herb species. These results were not tested for statistical significance. Over five years, mown and unmown plots contained distinct plant communities (data reported as a graphical analysis). In mown plots, there were 15–18 species/plot after one mow but 14–24 species/plot after five years of mowing (unmown plots stable at 5–9 species/plot). Total bryophyte cover was 5–15% after one mow but 23–75% after five years (data for unmown plots not reported). Cover of bindweed *Calystegia sepium* declined in mown plots only (from 11–51% to 1–29%; unmown plots stable at 31–62% cover). Cover of purple small-reed *Calamagrostis canescens* and common reed *Phragmites australis* declined in both mown and unmown plots. Three pairs of 100–200 m^2 plots were established in areas of partially drained, overgrown fen. Every winter between 1977/1978 and 1981/1982, one plot per pair was mown. The other plots were not mown. Each summer between 1978 and 1982, cover of every plant species was estimated in permanent quadrats (size and number not reported).

A replicated before-and-after study in 1978–1986 in a degraded fen in Belgium (3) reported that following the reinstatement of summer mowing, the plant community composition changed, species richness increased and cover of dominant herbs decreased. These results were not tested for statistical significance. Over eight years, the overall plant community composition in mown plots changed (data reported as a graphical analysis). There were 12–15 plant species/plot before mowing but 23–32 species/plot after seven years of mowing. Cover of the dominant herb species decreased: common reed *Phragmites australis* in two of three plots (from 66–82% to 14–30%), purple small-reed *Calamagrostis canescens* in two of three plots (from 51–80% to 24–27%), bindweed *Calystegia sepium* in all three plots (from 17–66% to 3–18%). Each summer between 1979 and 1985, three plots (200–300 m^2) in areas of partially drained, overgrown fen were mown once or twice. Cuttings were removed. Each summer between 1978 and 1986, cover of every plant species was estimated in permanent quadrats (size and number not reported).

A replicated, paired, site comparison study in 1998 in 27 fen meadows in Switzerland (4) found that mown meadows had greater plant species richness and vegetation cover than abandoned meadows, but shorter vegetation with less biomass. Mown meadows contained more plant species than abandoned meadows (mown: 33; abandoned: 27 species/8 m^2) and more fen-characteristic species (mown: 16; abandoned: 14 species/8 m^2). Plant diversity was also higher in mown meadows (reported as a diversity index). Mown meadows had greater cover of total vegetation (mown: 84%; abandoned: 77%) and mosses (mown: 47%; abandoned: 30%). Vegetation was shorter in mown meadows (mown: 16; abandoned: 24 cm) and total vegetation biomass was lower (mown: 265; abandoned: 320 g/cm^2). However, mown meadows contained greater sedge/rush biomass (mown: 146; abandoned: 102 g/cm^2). In summer 1998, vegetation was studied in 27 fen meadows: seven mown (each autumn for at least 20 years) and twenty abandoned (not mown for 2–35 years). Each mown meadow was matched with nearby abandoned meadows. Plant species and cover were recorded in four 2 m^2 plots/meadow. Above-ground biomass was cut in two 340 cm^2 quadrats/plot, then dried and weighed.

A replicated, randomized, controlled study in 1996–1998 in a degraded fen meadow in Germany (5) found that repeatedly clipped plots contained more species-rich and diverse vegetation than unclipped plots, and had a different community with more fen-characteristic plants. Over three years, plant species richness was significantly higher in clipped plots (17–23 species/2 m^2) than unclipped plots (15–18 species/2 m^2). Plant diversity was also higher in clipped plots (data reported as a diversity index). Overall plant community composition was initially similar in all plots but diverged over time. In clipped plots, shorter herbaceous species characteristic of fens and wet meadows became more abundant, taller sedges and reeds less so (data reported as a graphical analysis; changes not tested for statistical significance). Twenty 2 m^2 plots were established in an abandoned fen meadow. In ten random plots, vegetation was manually clipped (5–10 cm above the ground) every summer between 1996 and 1998. Ten plots were unclipped controls. Litter was removed from half of the plots in each treatment. Cover of every plant species was estimated annually, after clipping, in each plot.

A replicated, paired, site comparison study in 32 fen meadows in southern Germany (6) found that mown meadows contained a significantly different plant community and more plant species than grazed meadows, but there was no difference in vegetation height or biomass. Meadows mown or grazed for at least 10 years had different overall plant communities (data reported as a graphical analysis). Mown meadows contained more plant species than grazed meadows, per meadow (mown 79; grazed: 71 species) and per 25 m^2 plot (mown: 51; grazed: 43 species), and more fen-characteristic species (mown: 19; grazed: 18 species/plot). Meadows did not differ significantly in vegetation height (mown: 24; grazed: 19 cm) or above-ground biomass (mown: 1,103; grazed: 954 g/m^2). Of the 32 studied meadows, 16 were mown each autumn and 16 were open to cattle (<0.5/ha) each summer. In August (year not reported), cover of every plant species was recorded in 25 m^2 plots: 51 across the mown meadows and 58 across the grazed meadows. Vegetation height was measured at three points in each meadow. Biomass was cut from three 25 x 25 cm quadrats then dried and weighed.

A replicated, randomized, paired, controlled study in 1998–2000 in 15 degraded fen meadows in Switzerland (7) found that resuming mowing increased plant species richness and bryophyte cover, but had no effect on other plant cover or biomass. After two years, species richness was higher in mown plots than unmown plots: of all plants (32 vs 28 species/2 m^2) and fen-characteristic plants (18 vs 16 species/2 m^2). Mown plots also had greater bryophyte cover than unmown plots (60 vs 47%). There were no significant differences in vascular plant cover (data not reported), total biomass (mown: 193; unmown: 225 g/m^2) or herb biomass (mown: 52–80; unmown: 51–110 g/m^2). Four 2 m^2 plots were established in each meadow (abandoned for 4–35 years). In September 1998 and 1999, two random plots in each meadow were mown. Cuttings were removed. The other two plots were not mown. In summer 2000, vegetation cover was visually estimated in each plot. Above-ground biomass from a 20 x 20 cm subplot was cut, dried and weighed.

A replicated, paired, controlled, before-and-after study in 2002–2007 in a degraded grassy fen in the Czech Republic (8) found that reinstating mowing changed plant community composition, increased vascular plant richness and increased bryophyte cover. In mown (but not unmown) plots, the overall plant community composition changed significantly over five years in favour of fen-characteristic plants (data reported as graphical analyses). Mown plots had higher vascular plant richness than unmown plots after four years (twice-mown: 16–18; once-mown: 13; unmown: 7–9 species/m^2) and greater bryophyte cover after two years (twice-mown: 85–95%; once-mown: 64–89%; unmown: 7–13%). Before intervention, all plots had similar vascular plant richness (6–9 species/m^2) and bryophyte cover (9–12%). Five blocks of three plots (2.5 x 2.5 m) were established in an abandoned fen, dominated by tall moor grass *Molinia arundinacea*. Between 2002 and 2007, one plot/block received each mowing treatment: none, mowing in September, or mowing in May and September. Each year before May mowing, cover of every plant species was estimated in a 1 m^2 quadrat in the centre of each plot.

A replicated, paired, site comparison study in 2005–2008 in a fen meadow in Poland (9) found that mown plots contained more plant species and more moss biomass than unmown plots, but similar total plant biomass. Plant species richness was higher in plots mown every year (25 species/210 sample pins) than plots not mown for about 15 years (21 species/210 sample pins). The most abundant species under both treatments were sedges: black sedge *Carex nigra* in mown plots (16% cover) and fibrous tussock sedge *Carex approprinquata* in unmown plots (23% cover). Mown plots contained greater moss biomass than unmown plots, but total plant biomass was similar under both treatments (reported as statistical model results). Between 2005 and 2008, fifteen pairs of 2 x 2 m plots were sampled in early July. In each pair, one plot was in mown fen meadow (mown in late summer for at least 30 years). The other plot was in abandoned fen meadow, not mown for approximately 15 years. In each plot, plant species touching 210 pins were recorded. Live above-ground vegetation was collected from a 0.25 m^2 quadrat, then dried and weighed.

A before-and-after, site comparison study in 2004–2009 in a degraded fen in Poland (10) found that in an area where mowing was resumed (also cleared of shrubs and rewetted), the plant community composition changed in favour of fen meadow and wet meadow species. Over five years, the overall community composition became more similar to target fen meadow vegetation (data reported as a graphical analysis; change not tested for statistical significance). The abundance of fen meadow and wet meadow plant species, including sedges *Carex* spp., significantly increased in the managed area but did not change in the target area (data reported as abundance indices). In 2004, annual late summer mowing was resumed in 0.7 ha of drained, overgrown fen. The area had been prepared by removing willow *Salix cinerea* shrubs, and was later rewetted. The study does not distinguish between the effects of these interventions. The managed area was compared to 0.9 ha of target, shrub-free, fen meadow vegetation (retained in depressions during the drained period, but also affected by the rewetting and mown every other year). Annually between 2004 (before intervention) and 2009, vegetation cover was estimated in 18–22 plots/area. Plots were 20 x 20 m.

A replicated, randomized, paired, controlled study in 2009–2011 in a degraded fen in Poland (11) found that mown and unmown plots had a similar total number of plant species, vegetation height, shrub cover and hollow-adapted moss cover after 1–2 years, but that mown plots had less cover of sedges *Carex* spp. (all data reported as standardised scores). Twelve pairs of plots were established in a historically drained, abandoned fen. In late summer 2009 or 2010, one random plot in each pair was mown using a modified snow groomer. The other plots were not disturbed. In 2011, cover of every plant species was estimated in 4 m^2 quadrats (number not reported).

A controlled, before-and-after study in 2000–2002 in a degraded fen meadow in Italy (12) found that after reinstatement of mowing, the biomass of common reed *Phragmites australis* decreased. After two years of mowing, reed biomass was lower in a plot mown twice each year (22 g/m^2) and a plot mown once each year (56 g/m^2) than in an unmown plot (130 g/m^2). Before intervention, reed biomass was similar in all plots (99–112 g/m^2). In July 2000, three 10 x 10 m plots were established in an area of abandoned fen meadow invaded by reeds. Reed shoots were counted and measured in three 1 m^2 quadrats/plot and reed biomass was calculated. Then, one plot was mown once each year (August 2000 and 2001), one was mown twice each year (February 2001 and 2002, plus August mowing), and one was not mown. In July 2002, biomass measurements were repeated.

A replicated before-and-after study in 2010–2013 in two degraded fens in Wales, UK (13) found that mowing (sometimes along with other interventions) reduced shrub and grass/sedge/rush cover, typically had no effect on cover of fen-characteristic species and plant richness/diversity, and had mixed effects on vegetation height. In five of six managed plots, there were declines in shrub cover (before intervention: 27–87%; after 2–3 years: 9–19%) and total grass/sedge/rush cover (before: 91–98%; after: 63–80%). Cover of purple moor grass *Molinia caerulea* decreased in three plots (before: 64–81%; after: 3–34%). In four or five plots, there was no change in cover of fen-characteristic mosses (<1% before and after), fen-characteristic herbs (before: <2%; after: <1%), plant species richness (before: 9–17; after: 8–14 species/4 m^2) or plant diversity (data reported as diversity indices). Management had mixed effects on vegetation height (increase: 1 plot; decrease: 2 plots; no change: 3 plots). Six 20 x 20 m plots were established across two abandoned fens. In autumn 2010 or spring 2011, each plot was mown once. Cuttings were removed. Four plots were also grazed. Two of these were also burned. The study does not distinguish between the effects of these interventions and mowing. Cover of every plant species was estimated before mowing (autumn 2010) and 2–3 years after (autumn 2012 or 2013), in five 4 m^2 quadrats/plot.

A replicated, randomized, paired, controlled, before-and-after study in 2011–2013 in three degraded fens in Wales, UK (14) found that mowing typically reduced cover of grass-like plants and shrubs, typically had no effect on bryophyte cover, forb cover and plant species richness, and had mixed effects on vegetation height. In three of four comparisons, mown plots had less cover than unmown plots of grasses/sedges/rushes overall (50–56% vs 71–94%), the dominant sedge species (19–30% vs 41–82%), and shrubs (7–24% vs 11–36%). In contrast, in three of four comparisons mown and unmown plots had similar bryophyte cover (0–2%), forb cover (3–15% vs 2–20%) and plant species richness (6–17 species/plot vs 7–16 species/plot). Mown plots contained shorter vegetation than unmown plots in two of four comparisons (for which mown: 69–74 cm; unmown: 99–104 cm). Before mowing, all plots had similar vegetation cover, species richness and vegetation height. Seventeen pairs of 10 x 10 m plots were established across three abandoned fens. In spring 2012, one random plot in each pair was mown once (with a mechanical mower or strimmer; cuttings were removed). The other plots were left unmown. Data were recorded before mowing (summer 2011) and 1–2 years after (summer 2012 and 2013), in five 4 m^2 quadrats/plot.

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(2) Gryseels M. (1989) Nature management experiments in a derelict reedmarsh. I: effects of winter cutting. *Biological Conservation*, 47, 171–193.
(3) Gryseels M. (1989) Nature management experiments in a derelict reedmarsh. II: effects of summer mowing. *Biological Conservation*, 48, 85–99.
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(5) Jensen K. & Meyer C. (2001) Effects of light competition and litter on the performance of *Viola palustris* and on species composition and diversity of an abandoned fen meadow. *Plant Ecology*, 155, 169–181.
(6) Stammel B., Kiehl K. & Pfadenhauer J. (2003) Alternative management on fens: response of vegetation to grazing and mowing. *Applied Vegetation Science*, 6, 245–254.
(7) Billeter R., Peintinger M. & Diemer M. (2007) Restoration of montane fen meadows by mowing remains possible after 4–35 years of abandonment. *Botanica Helvetica*, 117, 1–13.
(8) Hájkova P., Hájek M. & Kintrová K. (2009) How can we effectively restore species richness and natural composition of a *Molinia* invaded fen? *Journal of Applied Ecology*, 46, 417–425.
(9) Opdekamp W., Beauchard O., Backx H., Franken F., Cox T.J.S., van Diggelen R. & Meire P. (2012) Effects of mowing cessation and hydrology on plant trait distribution in natural fen meadows. *Acta Oecologica*, 39, 117–127.
(10) Kotowski W., Dzierża P., Czerwiński M., Kozub Ł. & Śnieg S (2013) Shrub removal facilitates recovery of wetland species in a rewetted fen. *Journal for Nature Conservation*, 21, 294–308.
(11) Kotowski W., Jabłońska E. & Bartoszuk H. (2013) Conservation management in fens: do large tracked mowers impact functional plant diversity? *Biological Conservation*, 167, 292–297.
(12) Fogli S., Brancaleoni L., Lambertini C. & Gerdol R. (2014) Mowing regime has different effects on reed stands in relation to habitat. *Journal of Environmental Management*, 134, 56–62.
(13) Birch K.S., Guest J.E., Shepherd S., Milner P., Jones P.S. & Hanson J. (2015) *Responses of Rich-Fen Annex I and Related Habitats to Restoration and Management Undertaken as part of the Anglesey & Lyn Fens LIFE Project*. Final Report of the Anglesey & Llyn Fens LIFE Project, Technical Report 7.
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8.6 Remove plant litter to maintain or restore disturbance

B F S

- **Two studies** examined the effect on peatland vegetation of removing plant litter to maintain or restore disturbance. One study was in fen meadow[1] and one was in a fen[2].
- **Plant community composition (2 studies):** Two replicated, controlled studies (one randomized, one paired, before-and-after) in a fen meadow in Germany[1] and a fen in Czech Republic[2] found that removing plant litter did not affect plant community composition.

- **Vegetation cover (1 study):** One replicated, paired, controlled, before-and-after study in a fen in the Czech Republic[2] found that removing plant litter did not affect bryophyte or tall moor grass cover.
- **Overall plant richness/diversity (2 studies):** One replicated, randomized, controlled study in a fen meadow in Germany[1] reported that removing plant litter increased plant species richness and diversity. However, one replicated, paired, controlled, before-and-after study in a fen in the Czech Republic[2] found that removing litter did not affect vascular plant diversity.

Background

Traditional management such as mowing, grazing and burning can prevent build up of plant litter (dead material). Some fens and fen meadows have been mown for hundreds of years to produce animal feed or bedding (Güsewell 2003). However, many historically cut peatlands have been abandoned over the past 60 years, especially those in remote areas (Middleton *et al.* 2006). Accumulation of litter may affect the growth of some or all plants by influencing temperature, light and nutrient availability (Weltzin *et al.* 2005). Litter also creates a physical barrier above growing seedlings and below any seeds that fall on top of it (Facelli & Pickett 1991).

Removing vegetation litter by hand may mimic some of the disturbance caused by traditional management. To be included as evidence in this section, studies must have examined the effect of litter removal alone (not, for example, the effect of removing litter from mown plots).

CAUTION: Disturbance is not desirable on all peatlands. Fen meadows and some fens may benefit from disturbance. Many other fens, bogs and peat swamps will not. In fact, accumulation of dead plant matter is fundamental to the process of peat formation.

Related intervention: prescribed burning, which will clear plant litter as part of its wider effects on peatland vegetation (Section 8.9).

Facelli J.M. & Pickett S.T.A. (1991) Plant litter: its dynamics and effects on plant community structure. *The Botanical Review*, 57, 1–32.

Güsewell, S. (2003) Management of *Phragmites australis* in Swiss fen meadows by mowing in early summer. *Wetlands Ecology and Management*, 11, 433–445.

Middleton B., Holsten B. & van Diggelen R. (2006) Biodiversity management of fens and fen meadows by grazing, cutting and burning. *Applied Vegetation Science*, 9, 307–316.

Weltzin J.F., Keller J.K., Bridgham S.D., Pastor J., Allen P.B. & Chen J. (2005) Litter controls plant community composition in a northern fen. *Oikos*, 110, 537–546.

A replicated, randomized, controlled study in 1996–1998 in a degraded fen meadow in Germany (1) reported that plots cleared of plant litter had higher plant species richness and diversity than plots where litter was not removed, but that community composition was similar under both treatments. These results are not based on tests of statistical significance. Over three years, litter-removal plots contained 18–19 plant species vs 15–16 in non-removal plots. Plant diversity was also higher in litter-removal plots (data reported as a diversity index). The overall plant community composition was initially similar in all plots, then changed over time but in a similar way in litter-removal and non-removal plots (data reported as a graphical analysis). In 1996, ten 2 m^2 plots were established in an abandoned fen meadow. In May 1996, 1997 and 1998, all dead plant material was removed from five random plots. Dead plant material was left in the other five plots. Cover of every plant species was estimated annually, after litter removal, in each plot.

A replicated, paired, controlled, before-and-after study in 2002–2007 in a degraded grassy fen in the Czech Republic (2) found that removing plant litter had no effect on community composition, richness of vascular plants, bryophyte cover or cover of dominant tall moor grass *Molinia arundinacea*. In plots where litter was removed, the overall plant community composition did not change significantly over five years (data reported as graphical analyses). There was also no change in richness of vascular plants (8–9 species/m^2 across all years) and fen-characteristic vascular plants (data not reported), bryophyte cover (9–33% across all years) and moor grass cover (data not reported). These measures also remained stable in plots where litter was not removed. In May 2002, five pairs of 2.5 x 2.5 m plots were established in an abandoned fen, dominated by tall moor grass. Each May until 2007, dead plant litter was raked from one plot/pair. Plant litter was left in the other plots. Each year before litter removal, cover of every plant species was estimated in a 1 m^2 quadrat in the centre of each plot.

(1) Jensen K. & Meyer C. (2001) Effects of light competition and litter on the performance of *Viola palustris* and on species composition and diversity of an abandoned fen meadow. *Plant Ecology*, 155, 169–181.
(2) Hájkova P., Hájek M. & Kintrová K. (2009) How can we effectively restore species richness and natural composition of a *Molinia* invaded fen? *Journal of Applied Ecology*, 46, 417–425.

8.7 Cut large trees/shrubs to maintain or restore disturbance B Ⓕ S

- **Two studies** examined the effect on peatland vegetation of cutting large trees/shrubs to maintain or restore disturbance. One study was in a forested fen[1] and one was in an open fen[2]. *N.B. Section 9.6 considers cutting large trees/shrubs in peatlands with no clear history of disturbance.*
- **Plant community composition (1 study):** One before-and-after, site comparison study in a fen in Poland[2] found that in an area where shrubs were removed (along with other interventions), the plant community composition became more like a target fen meadow.
- **Characteristic plants (1 study):** One before-and-after, site comparison study in a fen in Poland[2] found that in an area where shrubs were removed (along with other interventions), the abundance of fen meadow plant species increased.
- **Vegetation cover (1 study):** One replicated, paired, controlled study in a forested fen in the USA[1] found that cutting and removing trees increased herb cover, but had no effect on shrub cover.
- **Vegetation structure (1 study):** One replicated, paired, controlled study in a forested fen in the USA[1] found that cutting and removing trees increased herb biomass and height.

Background

Regular disturbance may maintain vegetation in a desirable, semi-natural state – particularly in fen meadows and some fens (Middleton 2012). In particular, disturbance can clear shrubs and trees that would otherwise become dominant. To control woody vegetation, conservationists may sometimes want to restore or compensate for a disturbance that has been lost. Initially, large trees and shrubs may need to be managed by cutting them close to the ground. Afterwards, a traditional mowing or grazing regime may be resumed to keep regrowth of trees or shrubs in check, and manage the vegetation as a whole (see Section 8.5).

CAUTION: Tree/shrub removal may be desirable on a subset of peatlands e.g. open bogs, fens and fen meadows. Tree thinning may be desirable on some naturally forested peatlands – but tree removal is more typically a threat here.

Related interventions: cutting/mowing herbaceous plants and small woody plants, to maintain or restore disturbance (Section 8.5) or control problematic plants whose growth is not attributed to loss of traditional management (Section 9.4).

Middleton B.A. (2012) Rediscovering traditional vegetation management in preserves: trading experiences between cultures and continents. *Biological Conservation*, 158, 750–760.

A replicated, paired, controlled study in 2002–2007 in a forested fen in New York State, USA (1) found that in areas where trees were felled and removed, herb cover, height and biomass were greater than in adjacent forested areas, whilst shrub cover was similar. After 4–5 years, cleared areas had greater cover than adjacent forested areas of forbs (66 vs 44%) and sedges (9 vs 3%). There was a similar, but non-significant, trend for cover of grass-like plants overall (cleared: 50%; forested: 34%) and ferns (cleared: 17%; forested: 9%). Shrub cover did not significantly differ between areas (cleared: 9%; forested: 10%). In cleared areas, herbs were taller overall (cleared: 44; forested: 25 cm) and produced more biomass (cleared: 68; forested: 21 g/0.25 m^2). In spring 2002 and 2003, all trees were cut and removed from 11 circular areas (5 m radius) in a forested fen. This mimicked historical human disturbance. For each cleared area, a forested area <40 m away provided a control. In August 2007, vegetation was surveyed in each area within nine 0.25 m^2 quadrats.

A before-and-after, site comparison study in 2004–2009 in a degraded fen in Poland (2) found that in an area cleared of shrubs (then rewetted and mown), the plant community composition changed in favour of fen meadow and wet meadow species. Over five years, the overall plant community composition in the managed area became more similar to target fen meadow vegetation (data reported as a graphical analysis; change not tested for statistical significance). The abundance of fen meadow and wet meadow species, including sedges *Carex* spp., increased in the managed area but did not change in the target area (data reported as abundance indices). In 2004, willow *Salix cinerea* shrubs were cleared from 0.7 ha of drained, overgrown fen. The area was then mown annually and rewetted. The study does not distinguish between the effects of these interventions. The managed area was compared to 0.9 ha of target, shrub-free, fen meadow vegetation (retained in depressions during the drained period, but also affected by the rewetting and mown every other year). Annually between 2004 (before shrub clearance) and 2009, cover of every plant species was estimated in 18–22 plots/area. Plots were 20 x 20 m.

(1) Scanga, S.E. & Leopold, D.J. (2012) Managing wetland plant populations: lessons learned in Europe may apply to North American fens. *Biological Conservation*, 148, 69–78.
(2) Kotowski W., Dzierża P., Czerwiński M., Kozub Ł. & Śnieg S (2013) Shrub removal facilitates recovery of wetland species in a rewetted fen. *Journal for Nature Conservation*, 21, 294–308.

8.8 Use grazing to maintain or restore disturbance

B F S

- **Four studies** examined the effect on peatland vegetation of using grazing to maintain or restore disturbance. All four studies were in fens or fen meadows. *N.B. Section 9.7 considers grazing in peatlands with no clear history of disturbance.*

- **Plant community composition (1 study):** One replicated, paired, site comparison study in Germany[1] found that the overall plant community composition differed between grazed and mown fen meadows.
- **Characteristic plants (3 studies):** One replicated, paired, controlled study in Germany[2] reported that the abundance of bog/fen-characteristic plants was similar in grazed and ungrazed fen meadows. One replicated before-and-after study in a fen in the UK[3] reported that cover of fen-characteristic mosses did not change after grazers were introduced. One replicated, paired, site comparison study, also in Germany[1], found that grazed fen meadows contained fewer fen-characteristic plant species than mown meadows.
- **Herb cover (2 studies):** Two before-and-after studies in fens in the UK[3,4], reported that grazing increased cover of some herb groups (cottongrasses[4], sedges[4] or all grass-like plants[3]). One of the studies[4] found that grazing reduced purple moor grass cover, but the other[3] found that grazing typically had no effect.
- **Moss cover (2 studies):** One replicated before-and-after study in a fen in the UK[3] reported that cover of fen-characteristic mosses did not change after grazers were introduced. One controlled, before-and-after study in a fen in the UK[4] found that grazing reduced *Sphagnum* moss cover.
- **Tree/shrub cover (2 studies):** Of two before-and-after studies in fens in the UK, one[4] found that grazing reduced shrub cover but the other[3] found that grazing typically had no effect on shrub cover.
- **Overall plant richness/diversity (3 studies):** Of two before-and-after studies in fens in the UK, one[4] found that plant species richness increased after grazing was reinstated but the other[3] reported that there was typically no effect. One replicated, paired, site comparison study in Germany[1] found that grazed fen meadows contained fewer plant species than mown meadows.

Background

Regular disturbance may maintain vegetation in a desirable, semi-natural state – particularly in fen meadows and some fens (Middleton 2012). Disturbance can clear dominant plants such as trees and shrubs, maintain light availability and control nutrient levels. This can favour a plant community rich in plant species and/or peatland characteristic species. Therefore, conservationists may sometimes want to actively maintain or restore disturbance. Grazing by large vertebrates (e.g. sheep or cows) could be one way to do this.

Grazing itself may be the disturbance that has been reduced. Some peatlands have been historically grazed, usually at low intensities. Over the last 60 years many have been abandoned, especially those in remote areas (Middleton *et al.* 2006).

CAUTION: Disturbance is not desirable on all peatlands. Fen meadows and some fens may benefit from disturbance but many other fens, bogs and peat swamps will not. Careful management of grazing (e.g. species used, density, timing) may be necessary.

Related interventions: use grazing to control problematic plants whose growth is not attributed to loss of traditional management (Section 9.7); reduce intensity of grazing (Section 3.7); change type of livestock (Section 3.8); change season/timing of grazing (Section 3.9).

Middleton B., Holsten B. & van Diggelen R. (2006) Biodiversity management of fens and fen meadows by grazing, cutting and burning. *Applied Vegetation Science*, 9, 307–316.

Middleton B.A. (2012) Rediscovering traditional vegetation management in preserves: trading experiences between cultures and continents. *Biological Conservation*, 158, 750–760.

A replicated, paired, site comparison study in 32 fen meadows in Germany (1) found that grazed meadows contained a significantly different plant community with fewer species than mown meadows, but there was no difference in vegetation height or biomass. Meadows grazed or mown for at least 10 years had different overall plant communities (data reported as a graphical analysis). Grazed meadows contained fewer plant species than mown meadows, per meadow (71 vs 79) and per 25 m^2 plot (43 vs 51), and fewer fen-characteristic plant species (18 vs 19 species/plot). Meadows did not differ significantly in vegetation height (grazed: 19; mown: 24 cm) or above-ground biomass (grazed: 954; mown: 1,103 g/m^2). Of the 32 studied meadows, 16 were open to cattle (<0.5/ha) each summer and 16 were mown each autumn. In August (year not reported), cover of every plant species was recorded in 25 m^2 plots: 58 across the grazed meadows and 51 across the mown meadows. Vegetation height was measured at three points in each meadow. Biomass was cut from three 25 x 25 cm quadrats then dried and weighed.

A replicated, paired, controlled study in 2001–2005 in a degraded fen meadow in Germany (2) reported that grazing had no effect on the abundance of bog/fen-characteristic plants. This result is not based on a test of statistical significance. After five years, bog/fen-characteristic plants occurred in 0–3% of quadrats with 0–3% cover in both grazed and ungrazed plots. Variation between plots was related to topsoil stripping rather than grazing. In 2001, sixteen 6 x 6 m plots were established, in four blocks of four, in a drained, abandoned, nutrient-enriched fen meadow. Eight plots (two plots/block) were grazed by cattle (1.5 cattle/ha). The other eight plots were fenced to exclude cattle. None of these plots were sown with hay, but topsoil was stripped from four grazed and four ungrazed plots at the start of the experiment. Between 2002 and 2005, vegetation cover was estimated in 16 permanent 1 m^2 quadrats/plot.

A replicated before-and-after study in 2011–2013 in a degraded fen in Wales, UK (3) found that grazing typically increased cover of grass-like plants but had no effect on other vegetation cover, diversity or structure. Cover of grasses/sedges/rushes significantly increased in two of three grazed plots (from 78–79% before grazing to 93–105% after 16–21 months of grazing). There was no significant change in two of three plots (but a decrease in the other) in purple moor grass *Molinia caerulea* cover, dwarf shrub cover, fen-characteristic moss cover, fen-characteristic herb cover, plant species richness, plant diversity or vegetation height. Three 10 x 10 m plots were established in an abandoned fen. From spring or summer 2012, each plot was grazed by cattle or ponies. Before grazing began (August 2011) and after 16–21 months (autumn 2013) measurements were taken in five 4 m^2 quadrats/plot. Cover of every plant species was estimated, and vegetation height was measured, in the centre of each quadrat.

A controlled, before-and-after study in 2003–2012 in a historically grazed and recently burned fen in England, UK (4) found that grazing increased plant species richness, but reduced total vegetation cover and had mixed effects on cover of individual plant groups. Over nine years, grazed plots experienced a greater increase in plant species richness, but a smaller increase in total vegetation cover, than ungrazed plots (data not reported). For some vegetation, such as common cottongrass *Eriophorum angustifolium* and carnation sedge *Carex panicea,* cover increased more in grazed than ungrazed plots. For other vegetation, cover increased less in grazed plots. This included soft bog moss *Sphagnum tenellum* (data not reported), purple moor grass *Molinia caerulea* (before: 14–48%; grazed after: 14–43%; ungrazed after: 15–

64%) and dwarf shrubs (before: 3–8%; grazed after: 22–37%; ungrazed after: 33–53%). In summer 2003, cover of every plant species was estimated in 174 permanent 1 m^2 quadrats across the fen. From 2005, summer-autumn cattle grazing was reinstated (see original paper for details) except in three fenced exclosures. In 2010 and 2012, vegetation cover was re-surveyed in the quadrats (116 grazed, 58 ungrazed).

(1) Stammel B., Kiehl K. & Pfadenhauer J. (2003) Alternative management on fens: response of vegetation to grazing and mowing. *Applied Vegetation Science*, 6, 245–254.
(2) Rasran L., Vogt K. & Jensen K. (2007) Effects of topsoil removal, seed transfer with plant material and moderate grazing on restoration of riparian fen grasslands. *Applied Vegetation Science*, 10, 451–460.
(3) Birch K.S., Guest J.E., Shepherd S., Milner P., Jones P.S. & Hanson J. (2015) *Responses of Rich-Fen Annex I and Related Habitats to Restoration and Management Undertaken as part of the Anglesey & Lyn Fens LIFE Project*. Final Report of the Anglesey & Llyn Fens LIFE Project, Technical Report 7.
(4) Groome G.M. & Shaw P. (2015) Vegetation response to the reintroduction of cattle grazing on an English lowland valley mire and wet heath. *Conservation Evidence*, 12, 33–39.

8.9 Use prescribed fire to maintain or restore disturbance B Ⓕ S

- **Three studies** examined the effects on peatland vegetation of using prescribed fire to maintain or restore disturbance. Two studies were in fens[1,3] and one was in a bog[2]. *N.B. Section 9.8 considers prescribed burning in peatlands with no clear history of disturbance.*
- **Characteristic plants (1 study):** One replicated before-and-after study in a fen in the UK[3] reported that burning (along with other interventions) had no effect on cover of fen-characteristic mosses or herbs.
- **Herb cover (2 studies):** One replicated, controlled study in a fen in the USA[1] reported that burning reduced forb cover and increased sedge/rush cover but had no effect on grass cover. In contrast, one replicated before-and-after study in a fen in the UK[3] reported that burning (along with other interventions) reduced grass/sedge/rush cover.
- **Tree/shrub cover (2 studies):** Two replicated studies in fens in the USA[1] and the UK[3] reported that burning (sometimes[3] along with other interventions) reduced tree/shrub cover.
- **Overall plant richness/diversity (3 studies):** Two replicated, controlled studies in a fen in the USA[1] and a bog in New Zealand[2] found that burning increased plant species richness or diversity. However, one replicated before-and-after study in a fen in the UK[3] reported that burning (along with other interventions) typically had no effect on plant species richness and diversity.

Background

Regular disturbance may maintain vegetation in a desirable, semi-natural state – particularly in fen meadows and some fens (Middleton 2012). Disturbance can clear dominant plants, maintain light availability and control nutrient levels. This can favour a plant community rich in plant species and/or peatland characteristic species. Therefore, conservationists may sometimes want to actively maintain or restore disturbance. Prescribed burns might be one way to do this.

Burning itself may be the disturbance that has been reduced, as a result of land abandonment or because landscape changes (such as roads or ditches) create fire breaks. Historically, some North American fens burned annually due to lightning fires or intentional burning by humans (Middleton *et al.* 2006). Burning on some bogs helps to clear dominant vegetation (Norton & De Lange 2003).

CAUTION: Disturbance is not desirable on all peatlands. Fen meadows and some fens may benefit from disturbance. Many other fens, bogs and peat swamps will not. Natural fires in bogs and tropical peat swamps are rare, occurring every few *centuries* (Lindsay *et al*. 2011; Page & Hooijer 2016). Further risks specific to prescribed fires include the difficulty of controlling their intensity, duration and area. Uncontrolled burns can damage seed banks, *Sphagnum* mosses and the peat itself. Prescribed burns may be best carried out in the winter, when the peat is cold and wet (sometimes frozen), to avoid setting the peat on fire and reduce the risk of fire spreading beyond the prescribed area. Also note that burning might produce apparently desirable changes in vegetation over the short term (e.g. less heather cover and increased herb cover), followed by a rapid return to a degraded state.

Related intervention: use prescribed fire to control problematic plants whose growth is not attributed to loss of traditional management (Section 9.8).

Middleton B.A. (2012) Rediscovering traditional vegetation management in preserves: trading experiences between cultures and continents. *Biological Conservation*, 158, 750–760.

Middleton B., Holsten B. & van Diggelen R. (2006) Biodiversity management of fens and fen meadows by grazing, cutting and burning. *Applied Vegetation Science*, 9, 307–316.

Lindsay R., Birnie R. & Clough J. (2011) *Burning*. IUCN UK Peatland Programme Briefing Note No. 8.

Norton D.A. & De Lange P.J. (2003) Fire and vegetation in a temperate peat bog: implications for the management of threatened species. *Conservation Biology*, 17, 138–148.

Page S.E. & Hooijer A. (2016) In the line of fire: the tropical peatlands of South East Asia. *Philosophical Transactions of the Royal Society B*, 371, 20150176.

A replicated, controlled study in 1998–2000 in a degraded, shrubby sedge meadow in Wisconsin, USA (1) found that burned plots contained more plant species than unburned plots, and had greater sedge/rush cover, but lower tree/shrub and forb cover and similar grass cover. Over two subsequent years, species richness was higher in burned plots (7.1 species/0.2 m^2) than unburned plots (6.5 species/0.2 m^2). Burned plots also had greater cover of sedges/rushes (burned: 15–39%; unburned: 10–28%), but lower tree/shrub cover (burned: 0–11%; unburned: 6–12%) and lower forb cover (burned: 11–28%; unburned: 18–35%). Grass cover was similar in burned (1–12%) and unburned plots (0–9%). The cover results were not tested for statistical significance. Fifty-six 20 x 20 m plots were established in a degraded sedge meadow (historically burned and, in parts, grazed). Sedge meadows are sedge-dominated peatlands, fed by ground water. In December 1998, 33 plots were burned whilst 23 were not. In August 1999 and 2000 cover and height of every species were recorded, in one 0.2 m^2 quadrat/plot.

A replicated, controlled, before-and-after study in 1994–1998 in a bog in New Zealand (2) found that burned plots contained a different plant community to unburned plots, with greater plant species richness, diversity and cover. Before intervention, all plots contained a similar overall plant community. After four years, burned and unburned plots contained different communities (data reported as a graphical analysis; difference not tested for statistical significance). Also, burned plots experienced significant increases in foliage cover (from 103% before burning to 171% four years after), plant species richness (from 8 to 14 species/4 m^2) and plant diversity (data reported as a diversity index). In unburned plots, these measures declined (cover: from 104 to 100%; richness: from 9 to 6 species/4 m^2). In July (winter) 1994, twelve 2 x 2 m plots in a fire-suppressed bog were burned. Twelve control plots remained unburned. Cover of every plant species was recorded in all plots immediately before burning, and at intervals until September 1998.

A replicated before-and-after study in 2010–2012 in a degraded fen in Wales, UK (3) reported that burning (along with other interventions) reduced grass/sedge/rush and shrub cover, but typically had no effect on fen-characteristic plant cover and overall diversity, and had mixed effects on vegetation height. In both managed plots, there were decreases in total grass/sedge/rush cover (before burning: 97–98%; two years after: 70–74%) and shrub cover (before: 7–81%; after: 10–13%). Cover of purple moor grass *Molinia caerulea* decreased significantly in one plot with a similar trend in the other (before: 4–64%; after: 0–3%). There was no significant change in cover of fen-characteristic mosses (<1% before and after), fen-characteristic herbs (before: <2%; after: <1%), or plant species richness/diversity (in three of four comparisons). Vegetation height decreased in one plot but did not change in the other. Two 20 x 20 m plots were established in an abandoned fen. In September 2011, both plots were burned. After burning, one plot was mown and both plots were lightly grazed by cattle. The study does not distinguish between the effects of these interventions and burning. Cover of every plant species was estimated before burning (August 2008) and two years after (autumn 2013), in five 4 m^2 quadrats/plot.

(1) Middleton B. (2002) Winter burning and the reduction of *Cornus sericea* in sedge meadows in southern Wisconsin. *Restoration Ecology*, 10, 723–730.
(2) Norton D.A. & De Lange P.J. (2003) Fire and vegetation in a temperate peat bog: implications for the management of threatened species. *Conservation Biology*, 17, 138–148.
(3) Birch K.S., Guest J.E., Shepherd S., Milner P., Jones P.S. & Hanson J. (2015) *Responses of Rich-Fen Annex I and Related Habitats to Restoration and Management Undertaken as part of the Anglesey & Lyn Fens LIFE Project*. Final Report of the Anglesey & Llyn Fens LIFE Project, Technical Report 7.

Interventions: Modified wild fire regime

8.10 Thin vegetation to prevent wild fires

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of thinning vegetation to prevent wild fires.

Background

Naturally, most peatlands do not burn often. Bogs and tropical peat swamps may only burn every few centuries (Lindsay *et al*. 2011; Page & Hooijer 2016). Some fens may burn more often, perhaps annually in parts of North America (Middleton *et al.* 2006).

Frequent, intense, uncontrolled wild fires may be damaging to peatland vegetation that is not adapted to cope with them. Such fires are becoming more common as a result of peatland drainage (so the peat becomes drier), logging (which opens up the canopy and makes the forest warmer and drier) and climate change (warmer, drier, more storms with lightning) (Turetsky *et al.* 2014; Page & Hooijer 2016). Removing some vegetation to reduce the amount of fuel for wild fires could reduce their frequency and intensity, limiting the damage to vegetation.

CAUTION: In peatland fires, the peat itself can burn. Fire can spread underground, within the peat. Reducing vegetation fuel loads may not control these risks.

Related interventions: rewet peat to prevent wild fires (Section 8.11); build fire breaks (Section 8.12); methods of controlling vegetation biomass: cutting, physical removal, herbicide and prescribed burning (Sections 8.5–8.9 and Chapter 9); increase 'on the ground' protection, including fire fighting teams (Section 14.5).

Lindsay R., Birnie R. & Clough J. (2011) *Burning*. IUCN UK Peatland Programme Briefing Note No. 8.

Middleton B., Holsten B. & van Diggelen R. (2006) Biodiversity management of fens and fen meadows by grazing, cutting and burning. *Applied Vegetation Science*, 9, 307–316.

Page S.E. & Hooijer A. (2016) In the line of fire: the tropical peatlands of South East Asia. *Philosophical Transactions of the Royal Society B*, 371, 20150176.

Turetsky M.R., Benscoter B., Page S., Rein G., van der Werf G.R. & Watts A. (2014) Global vulnerability of peatlands to fire and carbon loss. *Nature Geoscience*, 8, 11–14.

8.11 Rewet peat to prevent wild fires

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of rewetting peat to prevent wild fires.

Background

Naturally, most peatlands do not burn often. Bogs and tropical peat swamps may only burn every few centuries (Lindsay *et al*. 2011; Page & Hooijer 2016). Some fens may burn more often, perhaps annually in parts of North America (Middleton *et al.* 2006).

Frequent, intense, uncontrolled wild fires may be damaging to peatland vegetation that is not adapted to cope with them. Along with logging and climate change, peatland drainage is a key factor increasing the amount of damaging fire in peatlands (Turetsky *et al.* 2014; Page & Hooijer 2016). Drained peatlands may be more likely to burn and can burn more intensely (deeper into the peat) when they do (Wösten *et al.* 2008). Raising the water table to rewet the surface peat, for example by building dams, may combat this threat.

Related interventions: rewetting alone, without a link to fire prevention (Section 8.1); clear/remove vegetation to prevent wild fire (Section 8.10); build fire breaks (Section 8.12); increase 'on the ground' protection, including fire fighting teams (Section 14.5).

Lindsay R., Birnie R. & Clough J. (2011) *Burning*. IUCN UK Peatland Programme Briefing Note No. 8.

Middleton B., Holsten B. & van Diggelen R. (2006) Biodiversity management of fens and fen meadows by grazing, cutting and burning. *Applied Vegetation Science*, 9, 307–316.

Page S.E. & Hooijer A. (2016) In the line of fire: the tropical peatlands of South East Asia. *Philosophical Transactions of the Royal Society B*, 371, 20150176.

Turetsky M.R., Benscoter B., Page S., Rein G., van der Werf G.R. & Watts A. (2014) Global vulnerability of peatlands to fire and carbon loss. *Nature Geoscience*, 8, 11–14.

Wösten J.H.M., Clymans E., Page S.E., Rieley J.O. & Limin S.H. (2008) Peat-water interrelationships in a tropical peatland ecosystem in Southeast Asia. *Catena*, 73, 212–224.

8.12 Build fire breaks

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of building fire breaks.

Background

Naturally, most peatlands do not burn often. Bogs and tropical peat swamps may only burn every few centuries (Lindsay *et al*. 2011; Page & Hooijer 2016). Some fens may burn more often, perhaps annually in parts of North America (Middleton *et al.* 2006). Frequent, intense, uncontrolled wild fires may be damaging to peatland vegetation that is not adapted to cope with them.

Fire breaks could be constructed to restrict fires to smaller areas of peatlands. Fire breaks could be strips of resistant trees, strips cleared of vegetation, embankments, empty ditches or water-filled ditches (Adinugroho *et al.* 2011). CAUTION: Some fire breaks are only suitable as short-term measures and should be dismantled once the fire risk has passed (e.g. ditches that can drain the peatland, Bess *et al.* 2014).

Related interventions: clear/remove vegetation to prevent wild fire (Section 8.10); rewet peat to prevent wild fire (Section 8.11); plant shelter belts to protect peatlands (Section 11.2); increase 'on the ground' protection, including fire fighting teams (Section 14.5).

Adinugroho W.C., Suryadiputra I.N.N., Saharjo B.H. & Siboro L. (2011) *Manual for the Control of Fire in Peatlands and Peatland Forest*. Wetlands International Indonesia & Wildlife Habitat Canada, Bogor.

Bess J.A., Chimner R.A. & Kangas L.C. (2014) Ditch restoration in a large Northern Michigan fen: vegetation response and basic porewater chemistry. *Ecological Restoration*, 32, 260–274.

Lindsay R., Birnie R. & Clough J. (2011) *Burning*. IUCN UK Peatland Programme Briefing Note No. 8.

Middleton B., Holsten B. & van Diggelen R. (2006) Biodiversity management of fens and fen meadows by grazing, cutting and burning. *Applied Vegetation Science*, 9, 307–316.

Page S.E. & Hooijer A. (2016) In the line of fire: the tropical peatlands of South East Asia. *Philosophical Transactions of the Royal Society B*, 371, 20150176.

8.13 Adopt zero burning policies near peatlands

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- We captured no evidence for the effect on peatland vegetation of adopting zero burning policies near peatlands.

Background

Fire is used, especially in tropical peatlands, to clear vegetation and prepare land for farming. If this fire is not controlled, it can spread to natural peatlands (Cattau *et al.* 2016). Land managers could adopt a zero burning policy in order to reduce the risk of fire escape. Procedures could involve cutting and shredding waste vegetation, then leaving it on site to decompose rather than burning it (Adinugroho *et al.* 2011). Zero burning policies have been adopted at the national level, and sometimes written into law, by members of the Association of South East Asian Nations (Adinugroho *et al.* 2011). Some large plantation companies have also adopted zero burning policies (Page & Hooijer 2016).

Related interventions: clear/remove vegetation to prevent wild fire (Section 8.10); rewet peat to prevent wild fire (Section 8.11); increase 'on the ground' protection, including fire fighting teams (Section 14.5); education and awareness-raising to prevent wild fire (Section 15.1).

Adinugroho W.C., Suryadiputra I.N.N., Saharjo B.H. & Siboro L. (2011) *Manual for the Control of Fire in Peatlands and Peatland Forest*. Wetlands International Indonesia & Wildlife Habitat Canada, Bogor.

Cattau M.E., Harrison M.E., Shinyo I., Tungau S., Uriarte M. & DeFries R. (2016) Sources of anthropogenic fire ignitions on the peat-swamp landscape in Kalimantan, Indonesia. *Global Environmental Change*, 39, 205–219.

Page S.E. & Hooijer A. (2016) In the line of fire: the tropical peatlands of South East Asia. *Philosophical Transactions of the Royal Society B*, 371, 20150176.

9. Threat: Invasive and other problematic species

Background

This chapter considers direct management of plants and animals that have harmful effects on biodiversity following their introduction, spread or increase in abundance.

In this synopsis, we define invasive species (in line with the International Union for the Conservation of Nature) as organisms that have been introduced to an area where they don't naturally occur and cause problems in this new range. This definition focuses on non-native or alien species, which tend to have more severe negative impacts than native species (Hassan & Ricciardi 2014).

However, native species can cause problems when they do not belong in a desired habitat or when they become overabundant. For example, trees are generally not desirable on open blanket bogs, whilst *Sphagnum* mosses are characteristic of bogs but not rich fens. Herbs and shrubs, including purple moor grass *Molinia caerulea*, are part of many natural peatland habitats but dominance is generally undesirable. With dense cover of one species, overall plant diversity can be reduced (Jensen & Meyer 2001). Controlling dominant plant species can create space for other species to grow. Controlling animal species can prevent them from causing damage, directly or indirectly, to peatland vegetation.

This chapter includes studies that control *succession of the whole plant community*, as well as studies targeting *specific plant or animal species*. We only include studies that report effects of controlling problematic species on the wider plant community i.e. studies that *only* report the effect of an intervention on the target problematic species are not included. Such studies are, or will be, summarized in other Conservation Evidence synopses.

The vegetation management interventions in this chapter are similar to those presented in Chapter 8 (mowing, cutting, grazing and prescribed fire). In Chapter 8, the interventions are used to maintain or restore historical disturbance. The current chapter considers using these interventions where there is no history of disturbance in a site. They might be used, for example, to tackle species that have become problematic due to other changes in site characteristics (such as drainage or increased nutrient levels).

Related threats: livestock and plants associated with agriculture, as opposed to wild problematic species (Chapter 3). Related interventions: interventions that manage the physical environment to make conditions less favourable for problematic species are considered under the relevant threat (e.g. rewetting in Chapter 8, reducing nutrient inputs in Chapter 10); interventions from this chapter used to maintain or restore disturbance (Chapter 8); topsoil removal/burial which can control problematic species (Chapter 12).

Hassan A. & Ricciardi A. (2014) Are non-native species more likely to become pests? Influence of biogeographic origin on the impacts of freshwater organisms. *Frontiers in Ecology and the Environment*, 12, 218–223.

Jensen K. & Meyer C. (2001) Effects of light competition and litter on performance of *Viola palustris* and on species composition and diversity of an abandoned fen meadow. *Plant Ecology*, 155, 169–181.

Key messages

All problematic species

9.1 *Implement biosecurity measures to prevent introductions of problematic species* **0 studies**

We captured no evidence for the effect, on peatland vegetation, of implementing biosecurity measures to prevent introductions of problematic species.

Problematic plants

9.2 *Physically remove problematic plants* **3 studies**

Characteristic plants: One replicated, randomized, controlled study in a fen in Ireland reported that cover of fen-characteristic plants increased after mossy vegetation was removed.

Herb cover: Three replicated, controlled studies in fens in the Netherlands and Ireland reported mixed effects of moss removal on herb cover after 2–5 years. Results varied between species or between sites, and sometimes depended on other treatments applied to plots.

Moss cover: One replicated, randomized, controlled study in a fen in Ireland reported that removing the moss carpet reduced total bryophyte and *Sphagnum* moss cover for three years. Two replicated, controlled, before-and-after studies in fens in the Netherlands reported that removing the moss carpet had no effect on moss cover 2–5 years later in wet plots, but reduced total moss and *Sphagnum* cover in drained plots.

Overall plant richness/diversity: One replicated, controlled, before-and-after study in a fen in the Netherlands reported that removing moss from a drained area increased plant species richness, but that there was no effect in a wet area.

9.3 *Physically damage problematic plants* **0 studies**

We captured no evidence for the effect on peatland vegetation of physically damaging problematic plants.

9.4 *Use cutting/mowing to control problematic herbaceous plants* **4 studies**

Plant community composition: Two replicated, randomized, paired, controlled, before-and-after studies in rich fens in Sweden found that mowing typically had no effect on the overall plant community composition. One controlled study in a fen meadow in the UK reported that mown plots developed different plant communities to unmown plots.

Characteristic plants: One replicated, randomized, paired, controlled, before-and-after study in a fen in Sweden found that mown plots contained more fen-characteristic plant species than unmown plots, although their cover did not differ significantly between treatments.

Vegetation cover: Of two replicated, randomized, paired, controlled, before-and-after studies in rich fens in Sweden, one found that mowing had no effect on vascular plant or bryophyte cover over five years. The other study reported that mowing typically increased *Sphagnum* moss cover and reduced purple moor grass cover, but had mixed effects on cover of other plant species.

Growth: One replicated, controlled, before-and-after study in a bog in Estonia found that clipping competing vegetation did not affect *Sphagnum* moss growth.

9.5 *Change season/timing of cutting/mowing* **2 studies**

Plant community composition: One replicated, randomized, paired, before-and after study in a fen meadow in the UK reported that changes in plant community composition over time were similar in spring-, summer- and autumn-mown plots. One study in a peatland in the Netherlands reported that summer- and winter-mown areas developed different plant community types.

Overall plant richness/diversity: One replicated, randomized, paired, before-and after study in a fen meadow in the UK found that plant species richness increased more, over two years, in summer-mown plots than spring- or autumn-mown plots.

9.6 *Use cutting to control problematic large trees/shrubs* — 2 studies

Plant community composition: Two studies (one replicated, controlled, before-and-after) in fens in the USA and Sweden reported that the plant community composition changed following tree/shrub removal, becoming less like unmanaged fens or more like undegraded, open fen.

Characteristic plants: One study in a fen in Sweden found that species richness and cover of fen-characteristic plants increased after trees/shrubs were removed.

Vegetation cover: One study in a fen in Sweden found that bryophyte and vascular plant cover increased after trees/shrubs were removed. One replicated, controlled, before-and-after study in fens in the USA found that removing shrubs, along with other interventions, could not prevent increases in shrub cover over time.

Overall plant richness/diversity: One study in a fen in Sweden found that moss and vascular plant species richness increased after trees/shrubs were removed. However, one replicated, controlled, before-and-after study in fens in the USA found that removing shrubs, along with other interventions, *prevented* increases in total plant species richness.

9.7 *Use grazing to control problematic plants* — 0 studies

We captured no evidence for the effect on peatland vegetation of using grazing to control problematic plants.

9.8 *Use prescribed fire to control problematic plants* — 6 studies

Moss cover: One replicated, paired, controlled study in bogs in Germany found that burning increased moss/lichen/bare ground cover in the short term (2–7 months after burning). Three replicated, paired studies in one bog in the UK found that moss cover (including *Sphagnum*) was higher in plots burned more often.

Herb cover: Four replicated, paired studies (two also controlled) in bogs in Germany and the UK examined the effect of prescribed fire on cottongrass cover. One found that burning had no effect on cottongrass cover after 2–7 months. One found that burning increased cottongrass cover after 8–18 years. Two reported that cottongrass cover was similar in plots burned every 10 or 20 years. The study in Germany also found that burning reduced purple moor grass cover after 2–7 months but had mixed effects, amongst sites, on cover of other grass-like plants and forbs.

Tree/shrub cover: Four replicated, paired studies (two also controlled) in bogs in Germany and one bog in the UK found that burning, or burning more often, reduced heather cover. Two replicated, controlled studies in the bogs in Germany and fens in the USA found that burning had no effect on cover of other shrubs. In the USA, burning was carried out along with other interventions.

Vegetation structure: One replicated, paired, controlled study in a bog in the UK found that plots burned more frequently contained more biomass of grass-like plants than plots burned less often, but contained less total vegetation, shrub and bryophyte biomass.

Overall plant richness/diversity: Two replicated, controlled studies in the fens in the USA and a bog in the UK found that burning reduced or limited plant species richness. In the USA, burning was carried out along with other interventions.

9.9 *Use covers/barriers to control problematic plants* — 0 studies

We captured no evidence for the effect on peatland vegetation of using covers or barriers to control problematic plants.

9.10 *Use herbicide to control problematic plants* **1 study**

Plant community composition: One replicated, controlled, before-and-after study in fens in the USA found that applying herbicide to shrubs, along with other interventions, changed the overall plant community composition.

Tree/shrub cover: The same study found that applying herbicide to shrubs, along with other interventions, could not prevent increases in shrub cover over time.

Overall plant richness/diversity: The same study found that applying herbicide to shrubs, along with other interventions, *prevented* increases in plant species richness.

9.11 *Introduce an organism to control problematic plants* **1 study**

Plant community composition: One controlled, before-and-after study in a fen meadow in Belgium found that introducing a parasitic plant altered the plant community composition.

Vegetation cover: The same study found that introducing a parasitic plant reduced cover of the dominant sedge but increased moss cover.

Overall plant richness/diversity: The same study found that introducing a parasitic plant increased overall plant species richness.

Problematic animals

9.12 *Exclude wild herbivores using physical barriers* **1 study**

Vegetation cover: One replicated, paired, controlled study in a fen meadow in Poland reported that the effect of boar- and deer exclusion on vascular plant and moss cover depended on other treatments applied to plots.

Vegetation structure: The same study reported that the effect of boar- and deer exclusion on total vegetation biomass depended on other treatments applied to plots.

Overall plant richness/diversity: The same study reported that the effect of boar- and deer exclusion on plant species richness depended on other treatments applied to plots.

9.13 *Control populations of wild herbivores* **0 studies**

We captured no evidence for the effect on peatland vegetation of controlling populations of wild herbivores.

Interventions: All problematic species

9.1 Implement biosecurity measures to prevent introductions of problematic species

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- We captured no evidence for the effect, on peatland vegetation, of implementing biosecurity measures to prevent introductions of problematic species.

Background

It is often cheaper and easier to prevent problematic species from being introduced to a site than trying to control them afterwards (Leung *et al.* 2002). This section includes all interventions aiming to directly prevent introductions of problematic species: from physical biosecurity measures like cleaning and drying equipment between sites, to legislative measures like banning the sale or ownership of problematic species.

CAUTION: Bans on sale or ownership of problematic species may encourage mass releases into the wild, from people concerned about their newly illegal organisms (Hulme 2015).

Related intervention: raise awareness amongst the public, including about problematic species and biosecurity (Section 15.1).

Hulme P. (2015) European Union: new law risks release of invasive species. *Nature*, 517, 21.

Leung B., Lodge D.M., Finnoff D., Shogren J.F., Lewis M.A. & Lamberti G. (2002) An ounce of prevention or a pound of cure: bioeconomic risk analysis of invasive species. *Proceedings of the Royal Society B*, 269, 2407–2413.

Interventions: Problematic plants

9.2 Physically remove problematic plants

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- **Three** studies examined the effect on peatland vegetation of removing problematic plants. All three studies were in fens.
- **Characteristic plants (1 study):** One replicated, randomized, controlled study in a fen in Ireland[3] reported that cover of fen-characteristic plants increased after mossy vegetation was removed.
- **Herb cover (3 studies):** Three replicated, controlled studies in fens in the Netherlands[1,2] and Ireland[3] reported mixed effects of moss removal on herb cover after 2–5 years. Results varied between species or between sites, and sometimes depended on other treatments applied to plots (i.e. drainage[1] or isolation from the surrounding bog[3]).
- **Moss cover (3 studies):** One replicated, randomized, controlled study in a fen in Ireland[3] reported that removing the moss carpet reduced total bryophyte and *Sphagnum* moss cover for three years. Two replicated, controlled, before-and-after studies in fens in the Netherlands[1,2] reported that removing the moss carpet had no effect on moss cover (after 2–5 years) in wet plots, but reduced total moss[1] and *Sphagnum*[1,2] cover in drained plots.
- **Overall plant richness/diversity (1 study):** One replicated, controlled, before-and-after study in a fen in the Netherlands[2] reported that moss removal increased plant species richness, but only in a drained area.

Background

This section considers complete physical removal of problematic plants i.e. pulling up or digging up entire plants, or scraping living vegetation from the peatland surface. By completely removing plants, including roots where applicable, immediate regrowth will be prevented (although long-term recolonization is possible).

Physical removal can precisely target individuals of problematic species. Alternatively, it can involve broad clearance of dominant vegetation that outcompetes desirable plants or changes the physical environment to an undesirable state (e.g. acidification and nutrient enrichment by moss carpets on fens; Bootsma *et al.* 2002).

Related interventions: remove or bury upper layer of peat or soil, which will also involve removal of existing plants (Sections 12.6 and 12.7).

Bootsma M.C., van den Broek T., Barendregt A. & Beltman B. (2002) Rehabilitation of acidified floating fens by addition of buffered surface water. *Restoration Ecology*, 10, 112–121.

A replicated, controlled, before-and-after study in 1989–1993 in a degraded floating fen in the Netherlands (1) reported that the effect of moss removal on vegetation cover after two years depended on whether plots were previously drained. These results were not tested for statistical significance. All plots were initially dominated by mosses (moss cover: 83–96%; herb cover: <1–2%). Of two drained plots, one cleared of moss developed herb cover after two years (moss cover: 0%; herb cover: 76%) whereas one from which moss was not removed remained dominated by mosses (total moss cover: 99%; *Sphagnum* cover: 64%; herb cover: 3%). In two undrained plots, moss removal had no effect on vegetation cover. Plots with and without moss removal developed similar vegetation cover (total mosses: 93–96%; *Sphagnum*: 53–62%; herbs: 1–3%). In 1991, the moss carpet was cleared from two 16 m^2 plots in an acidified, nutrient-enriched fen. Two adjacent plots were not cleared. One cleared and one uncleared plot were also drained (by a ditch dug in 1989). In 1991 (before moss removal) and 1993, vegetation cover was recorded in six 1 m^2 quadrats/plot.

A replicated, controlled, before-and-after study in 1991–1996 in a degraded floating fen in the Netherlands (2) reported that moss removal consistently reduced cover of black sedge *Carex nigra* and common cottongrass *Eriophorum angustifolium*, but that the effect on plant species richness and *Sphagnum* moss cover depended on whether plots had been drained. These results were not tested for statistical significance. After five years, black sedge and common cottongrass were less abundant in plots cleared of moss (sedge: absent; cottongrass: in 21–60% of quadrats) than uncleared plots (sedge: in 11–20% of quadrats; cottongrass: in 61–100% of quadrats). In a drained area, moss removal increased the number of plant species/plot (removal: 32–43; non-removal: 22–36) and *Sphagnum* cover (removal: 61–100%; non-removal: 41–60%). However, in an undrained area, moss removal reduced the number of species/plot (removal: 14; non-removal: 16) and had no effect on *Sphagnum* cover (removal: 21–40%; non-removal: 21–40%). Before intervention, plots contained 16 species and had 21–40% *Sphagnum* cover. In January 1992, the thick moss carpet was cleared from six plots in an acidified, nutrient-enriched fen. Six adjacent plots were not cleared. Half of the plots were in an area drained of acidic surface water and half in an undrained area. Between 1991 and 1996, cover of every plant species was estimated in quadrats covering 6–8 m^2 of each plot.

A replicated, randomized, controlled study in 2003–2006 in a soak (fen-like part of a bog) in Ireland (3) reported that plots cleared of floating vegetation developed low cover of open water plant communities (if completely isolated from the surrounding bog) or high cover of fen-characteristic species (if partially isolated). No statistical tests were carried out. Three years after vegetation removal, completely isolated plots had 35% vegetation cover, comprised entirely of aquatic herbs (bryophyte, cottongrass *Eriophorum angustifolium* and sedge *Carex rostrata* cover all 0%). In partially isolated plots, vegetation cover was 85% (including wetland herbs: 40%; all bryophytes: 70%; *Sphagnum* moss 11%; cottongrass: 3%; sedge: 4%). In control plots from which vegetation was not removed, vegetation cover was 100% (including wetland herbs: 50%; all bryophytes: 95%; *Sphagnum* moss: 13%; cottongrass 3%; sedge: 3%). In October 2003, six 4 x 4 m plots were established. Floating peat and vegetation were removed from four random plots, of which two were then partially isolated from the surrounding bog (with porous plastic membranes) and two completely isolated (with impermeable rubber membranes). Two plots were not manipulated. In July 2004–2006, cover of every plant species was estimated in each plot.

(1) Beltman B., van den Broek T., Bloemen S. & Witsel C. (1996) Effects of restoration measures on nutrient availability in a formerly nutrient-poor floating fen after acidification and eutrophication. *Biological Conservation*, 78, 271–277.
(2) Bootsma M.C., van den Broek T., Barendregt A. & Beltman B. (2002) Rehabilitation of acidified floating fens by addition of buffered surface water. *Restoration Ecology*, 10, 112–121.
(3) Crushell P.H., Smolders A.J.P., Schouten M.G.C., Robroek B.J.M., van Wirdum G. & Roelofs J.G.M. (2011) Restoration of a terrestrialized soak lake of an Irish raised bog: results of field experiments. *Restoration Ecology*, 19, 261–272.

N.B. Results from (1) are also reported in: Beltman B., van den Broek T. & Bloemen S. (1995) Restoration of acidified rich-fen ecosystems in the Vechtplassen area: successes and failures. Pages 274–286 in: B.D. Wheeler, S.C. Shaw, W.J. Fojt & R.A. Robertson (eds.) *Restoration of Temperate Wetlands*. John Wiley & Sons Ltd., Chichester.

9.3 Physically damage problematic plants

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- We captured no evidence for the effect on peatland vegetation of physically damaging problematic plants.

Background

This section considers physically damaging problematic plants in order to control them. Damage may kill plants directly, increase their susceptibility to disease, slow their growth and/or prevent reproduction. This section includes damage done specifically to the plants (e.g. by crushing or seed head removal) as well as general soil disturbance that damages plants growing in it (see also Section 12.9). CAUTION: To avoid regrowth, it may be necessary to remove plant fragments. Also, damaging plants by disturbing peat can destroy the physical structure of the peat.

Related interventions: remove, bury or disturb peatland surface (Sections 12.6–12.8).

9.4 Use cutting/mowing to control problematic herbaceous plants

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- **Four studies** examined the effect on peatland vegetation of cutting/mowing problematic herbaceous plants. Three studies were in fens or fen meadows[1,3,4] and one was in a bog[2]. *N.B. Section 8.5 considers cutting/mowing in historically disturbed peatlands*.
- **Plant community composition (3 studies):** Two replicated, randomized, paired, controlled, before-and-after studies in rich fens in Sweden[3,4] found that mowing typically had no significant effect on the overall plant community composition. One controlled study in a fen meadow in the UK[1] reported that mown plots developed different plant communities to unmown plots.
- **Characteristic plants (1 study):** One replicated, randomized, paired, controlled, before-and-after study in a fen in Sweden[4] found that mown plots contained more fen-characteristic plant species than unmown plots, although their cover did not differ significantly between treatments.
- **Vegetation cover (2 studies):** Of two replicated, randomized, paired, controlled, before-and-after studies in rich fens in Sweden, one[4] found that mowing had no effect on vascular plant or bryophyte cover over five years. The other[3] reported that mowing typically increased *Sphagnum* moss cover and reduced purple moor grass cover, but had mixed effects on cover of other species.
- **Growth (1 study):** One replicated, controlled, before-and-after study in a bog in Estonia[2] found that clipping competing vegetation did not affect *Sphagnum* moss growth.

Background

Cutting or mowing is the removal of above-ground parts of herbaceous plants or young trees/shrubs. Roots are left in place. Mowing and cutting can be broad tools affecting all plants in a community, so are often used to manage succession (development of plant communities over time). Cuttings may be removed from the site or left in place to rot down. Note that this choice will affect nutrients, temperature and light within the peatland, in turn determining which plant species can grow (Weltzin *et al.* 2005; see also Section 8.6).

CAUTION: Mowing with heavy machinery could damage the peatland surface and vegetation. Cutting by hand or with specialized vehicles might cause less damage.

Related interventions: cutting/mowing to control herbaceous plants as part of a traditional disturbance regime (Section 8.5); change season of cutting/mowing (Section 9.5); use low impact vehicles or harvesting techniques (Sections 6.3 and 6.4).

Weltzin J.F., Keller J.K., Bridgham S.D., Pastor J., Allen P.B. & Chen J. (2005) Litter controls plant community composition in a northern fen. *Oikos*, 110, 537–546.

A controlled study in 1927–1940 in a fen meadow in England, UK (1) reported that repeated cutting changed the composition of the plant community. These results were not tested for statistical significance. After 12 years, a plot cut every year had a plant community dominated by purple moor grass *Molinia caerulea* with abundant carnation sedge *Carex panicea* (data reported as abundance categories). Sawtooth sedge *Cladium mariscus* biomass decreased over time (from 490 g/m^2 after one cut to 50 g/m^2 after 12 cuts). In an uncut plot, sawtooth sedge remained the most abundant plant species (data reported as abundance categories). Additional plots cut every two, three or four years developed plant communities intermediate between the annually cut and uncut plots. In 1927, five 20 x 20 m plots were established in a fen meadow dominated by sawtooth sedge. Four plots were scythed in October: one every year, one every two years, one every three years and one every four years. Cuttings were removed. The other plot was left uncut. Vegetation cover was visually estimated in 1940. Above-ground vegetation biomass was estimated every year, by drying and weighing cuttings from 1 m^2 of each plot.

A replicated, controlled, before-and-after study in 2003–2006 in a raised bog in Estonia (2) found that clipping competing plants did not significantly affect growth of Magellan's bog moss *Sphagnum magellanicum* (data not reported). In six plots dominated by Magellan's bog moss, vascular plants were clipped flush to the moss surface every May and September. Plants were not clipped in the six other plots. The height increase of Magellan's bog moss was measured each summer.

A replicated, randomized, paired, controlled, before-and-after study in 2002–2005 in two degraded rich fens in Sweden (3) reported that repeated mowing altered the plant community composition, reduced cover of purple moor grass *Molinia caerulea* and increased overall cover of *Sphagnum* moss. The cover results were not tested for statistical significance. Mowing altered the development of the overall plant community over three years, although only significantly so in one fen (data reported as a graphical analysis). In three of four comparisons, mown plots had lower cover than unmown plots of purple moor grass (mown: 1–37%; not mown: 2–50%) but higher overall cover of *Sphagnum* moss (mown: 2–41%; not mown: 1–28%). However, cover of individual *Sphagnum* species showed mixed responses to mowing amongst sites or other treatments applied to plots. The same was true for sedges *Carex* spp., common cottongrass *Eriophorum angustifolium* and common reed *Phragmites australis*. In autumn 2002, sixty-four 2.5 x 2.5 m plots were established (in four blocks of 16) across two degraded fens. Thirty-two plots (eight random plots/block) were mown every autumn between 2003 and 2005. Cuttings were removed. The other plots were not mown. Additionally, trees had been removed from all plots and some plots had been rewetted or dug over. In 2002 (before intervention) and 2005, cover of every plant species was estimated in one 0.25 m^2 quadrat/plot.

A replicated, randomized, paired, controlled, before-and-after study in 1996–2001 in a degraded rich fen in Sweden (4) found that mown and unmown plots maintained a similar overall plant community and vegetation cover, but that mown plots developed greater plant species richness. The overall plant community composition changed over five years, but in a similar way in mown and unmown plots (data reported as a graphical analysis). Likewise, vegetation cover increased by similar amounts, from similar initial values, in mown and unmown plots. This was true for vascular plants, fen-characteristic plants, bryophytes, six of eight *Carex* sedge species, common cottongrass *Eriophorum angustifolium*, purple moor grass *Molinia arundinacea* and common reed *Phragmites australis*. After five years, mown plots contained more vascular plant species than unmown plots (26 vs 18 species/m^2) and more fen-characteristic plant species (14 vs 10 species/m^2). Before mowing, species richness was similar in all plots (vascular: 15; fen-characteristic: 7–8). In 1996, nine pairs of 9 m^2 plots were established in a degraded fen. Every August until 2001 vegetation was cut by hand (and cuttings removed) in one random plot/pair. The other plots were not cut. All plots had been cleared of trees and shrubs and were grazed every summer (approximately 50 cows/ha). In 1996 (before mowing) and 2001, cover of every plant species was estimated in one 1 m^2 quadrat/plot.

(1) Godwin H. (1941) Studies in the ecology of Wicken Fen: IV. Crop-taking experiments. *Journal of Ecology*, 29, 83–106.
(2) Robroek B.J.M., van Ruijven J., Schouten M.G.C., Breeuwer A., Crushell P.H., Berendse F. & Limpens J. (2009) *Sphagnum* reintroduction in degraded peatlands: the effects of aggregation, species identity and water table. *Basic and Applied Ecology*, 10, 697–706.
(3) Mälson K., Sundberg S. & Rydin H. (2010) Peat disturbance, mowing, and ditch blocking as tools in rich fen restoration. *Restoration Ecology*, 18, 469–478.
(4) Sundberg S. (2011) Quick target vegetation recovery after restorative shrub removal and mowing in a calcareous fen. *Restoration Ecology*, 20, 331–338.

9.5 Change season/timing of cutting/mowing

B F S

- **Two studies** compared the effect on peatland vegetation of mowing or cutting in different seasons. One study was in a fen meadow[1] and one was in a peatland with mixed vegetation[2].
- **Plant community composition (2 studies):** One replicated, randomized, paired, before-and after study in a fen meadow in the UK[1] reported that changes in plant community composition were typically similar in spring-, summer- and autumn-mown plots. However, one study in a peatland in the Netherlands[2] reported that summer- and winter-mown areas developed cover of different plant community types.
- **Overall plant richness/diversity (1 study):** One replicated, randomized, paired, before-and after study in a fen meadow in the UK[1] found that plant species richness increased more, over two years, in summer-mown plots than spring- or autumn-mown plots.

Background

Studies in this section compare the effects of cutting/mowing at different times of year, given a fixed cutting frequency (i.e. number of cuts each year). Mowing could have different effects on peatland vegetation depending on the season in which it is done. For example, if an undesirable plant is only present above ground in some seasons, mowing must be done at the right time to damage this plant. The timing of cutting will also affect nutrient removal. Usually, more nutrients will be removed by cutting in summer or autumn: before the above-ground biomass of annual plants dies off and nutrients are stored in underground organs. In some peatlands, mowing has shifted from the traditional season (Rowell *et al.* 1985).

Related interventions: maintain or restore cutting/mowing as part of a traditional disturbance regime (Section 8.5); use cutting/mowing to manage vegetation: overall effect (Section 9.4).

Rowell T.A., Guarino L. & Harvey H.J. (1985) The experimental management of vegetation at Wicken Fen, Cambridgeshire. *Journal of Applied Ecology*, 22, 217–227.

A replicated, randomized, paired, before-and-after study in 1980–1982 in a fen meadow in England, UK (1) reported that mowing in different seasons generally produced similar changes in plant community composition, but found that summer-mown plots experienced larger increases in plant species richness than spring- and autumn-mown plots. Over two years, changes in overall plant community composition were similar in spring-, summer- and autumn-mown plots in four of six cases. In one plant community, autumn-mown plots developed different plant communities to spring- and summer-mown plots (data reported as graphical analyses; results not tested for statistical significance). Species richness increased significantly more in summer-mown plots than spring- or autumn-mown plots in three of four cases (summer-mown: 0.8–1.4 extra species; spring-mown: 0.2–0.5 extra species; autumn-mown: 0.2–0.3 extra species/250 cm^2). In 1980–1982, annual mowing treatments were randomly applied to four 25 m^2 plots in each of 10 blocks (situated in two different fen meadow community types). In each block, one plot was spring-mown (May), two were summer-mown (July) and one was autumn-mown (September). Cuttings were removed. Immediately before each mowing, vascular plant species were recorded in sixteen 250 cm^2 quadrats/plot.

A study in 1956–1989 in a historically mined peatland in the Netherlands (2) reported that summer- and winter-mown areas developed different types of plant communities. No statistical tests were carried out. Initially, areas destined for each mowing regime contained similar vegetation types: 42–52% of the surface was covered by reedbeds, 28–30% by fen vegetation (mostly alkaline 'rich' fens), 20–25% by meadows and 0% by bogs. After approximately 30 years of mowing, summer-mown areas had developed into acidic poor fens (62%) and bogs (21%), with some reedbeds (14%). In contrast, winter-mown areas had mainly developed into reedbeds (68%) and poor fens (32%). In 1989, vegetation was mapped in 5 ha of summer-mown peatland and 30 ha of winter-mown peatland. This was compared to maps created in 1956. Vegetation was developing on pools created by historical peat extraction. By 1989 the peatland had been mown for approximately 30 years, but it was not clear whether the peatland was abandoned or mown before this.

(1) Rowell T.A., Guarino L. & Harvey H.J. (1985) The experimental management of vegetation at Wicken Fen, Cambridgeshire. *Journal of Applied Ecology*, 22, 217–227.

(2) van Diggelen R., Molenaar W.J. & Kooijman A.M. (1996) Vegetation succession in a floating mire in relation to management and hydrology. *Journal of Vegetation Science*, 7, 809–820.

9.6 Use cutting to control problematic large trees/shrubs

Ⓑ Ⓕ S

- **Two studies** examined the effect on peatland vegetation of cutting and removing problematic large trees/shrubs. Both studies were in fens. *N.B. Section 8.7 considers cutting trees/shrubs in historically disturbed peatlands.*
- **Plant community composition (2 studies):** Two studies (one replicated, controlled, before-and-after) in fens in the USA[1] and Sweden[2] reported that the plant community composition changed following tree/shrub removal, becoming less like unmanaged fens[1] or more like undegraded, open fen[2].
- **Characteristic plants (1 study):** One study in a fen in Sweden[2] found that species richness and cover of fen-characteristic plants increased following tree/shrub removal.
- **Vegetation cover (2 studies):** One study in a fen in Sweden[2] found that bryophyte and vascular plant cover increased following tree/shrub removal. One replicated, controlled, before-and-after study in fens in the USA[1] found that shrub removal (along with other interventions) could not prevent increases in shrub cover over time.
- **Overall plant richness/diversity (2 studies):** One study in a fen in Sweden[2] found that moss and vascular plant species richness increased following tree/shrub removal. However, one replicated, controlled, before-and-after study in fens in the USA[1] found that shrub removal (along with other interventions) *prevented* increases in total plant species richness.

Background

This section considers cutting and the removal of above-ground parts of mature shrubs and trees: plants that are too large to mow. Roots are left in place. Cuttings may be removed from the site (which means the nutrients they contain are also removed), or left in place to rot down.

CAUTION: Tree/shrub removal may be desirable on a subset of peatlands e.g. open bogs, fens and fen meadows. Tree thinning may be desirable on some naturally forested peatlands – but tree removal is more typically a threat here.

Related interventions: cutting and removing trees that have grown as part of forestry operations (Sections 3.3 and 3.4); cutting to control trees/shrubs as part of a traditional disturbance regime (Section 8.7); completely removing problematic plants by digging/pulling (Section 9.2).

A replicated, controlled, before-and-after study in 1986–2000 in two shrub-invaded fens in Ohio, USA (1) found that cutting shrubs (along with burning and herbicide application) altered plant community composition and prevented increases in plant species richness, but had no effect on shrub cover. The overall plant community composition changed significantly over time along transects with and without shrub control, but they accumulated different sets of species (data reported as a graphical analysis). Plant species richness was stable in the fen with shrub control (before: 12.8; after 14 years: 12.7 species/m^2) but increased in the fen without shrub control (before: 12.5; after 14 years: 14.6 species/m^2). Woody plant cover increased similarly in fens with shrub control (before: 46%; after 11 years: 62%) and without shrub control (before: 20%; after: 28%). From 1986, encroaching shrubs were managed using *ad hoc* cutting, burning and herbicide application. The study does not distinguish between the effects of these interventions. Three of four transects were managed in one fen ('with shrub control'). Only one of four transects were managed in the other fen ('without shrub control'). In summer 1986 (before shrub control began), 1999 and 2000, vegetation cover was estimated in 1 m^2 quadrats along the eight transects. Shrub cover was estimated from aerial photographs.

A site comparison study in 1995–2001 in an overgrown rich fen in Sweden (2) found that following tree/shrub removal, the plant community composition became more like a natural fen, and plant species richness and vegetation cover increased. Between one and six years after shrub removal, the overall plant community composition became more like an open fen (data reported as a graphical analysis). Where shrubs were removed, species richness increased for vascular plants (from 15 to 18 species/m^2), bryophytes (from 7 to 9 species/m^2) and fen-characteristic plants (from 8 to 10 species/m^2). Cover of these groups also increased (vascular plants: from 18 to 24%; bryophytes: from 9 to 31%; fen-characteristic plants: from 7 to 15%), as did cover of common cottongrass *Eriophorum angustifolium* (from 0.3 to 0.6%) and three of eight *Carex* sedge species. Cover of five other sedge species, purple moor grass *Molinia caerulea* and common reed *Phragmites australis* did not change. In 1995, shrubs (mainly juniper *Juniperus communis*) and trees (conifers) were manually cut and removed from a 30 x 50 m area of overgrown fen. The fen was grazed by 7–12 cows every summer, both before and after shrub removal. Cover of every plant species was estimated in August 1996 and 2000: in nine 1 m^2 quadrats across the managed area and three quadrats in another part of the fen that had not become overgrown.

(1) Barry M.J., Barbara A.K. & De Szalay F. (2008) Long-term plant community changes in managed fens in Ohio, USA. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 18, 392–407.
(2) Sundberg S. (2011) Quick target vegetation recovery after restorative shrub removal and mowing in a calcareous fen. *Restoration Ecology*, 20, 331–338.

9.7 Use grazing to control problematic plants Ⓑ Ⓕ S

- We captured no evidence for the effect on peatland vegetation of using grazing to control problematic plants. *N.B. Sections 3.5, 3.6 and 8.8 consider grazing in different contexts.*

Background

This section considers using grazing vertebrates (e.g. sheep or cows) to control problematic plants. Grazers remove shoots or flowers, limiting plant growth and/or reproduction. They might selectively graze certain plant groups or species (Grant *et al.* 1987), creating space for other species to grow. CAUTION: Trampling, erosion and nutrient enrichment from grazers can have negative impacts on peatlands, especially if the density of grazers is high.

Related interventions: interventions to address domestic livestock as a threat, such as exclusion (Sections 3.5–3.9); grazing to manage plants as part of a traditional disturbance regime (Section 8.8).

Grant S.A., Suckling S.A., Smith H.K., Torvell L., Forbes T.D.A. & Hodgson J. (1987). Comparative studies of diet selection by sheep and cattle: blanket bog and heather moor. *Journal of Ecology*, 75, 947–960.

9.8 Use prescribed fire to control problematic plants Ⓑ Ⓕ S

- **Six studies** examined the effect on peatland vegetation of using prescribed fire to control problematic plants: five in bogs[1,2,3,4,6] and one in fens[5]. Four studies[1,2,4,6] were based on the same experimental set-up in the UK. *N.B. Section 8.9 considers prescribed burning in historically disturbed peatlands.*
- **Moss cover (4 studies):** One replicated, paired, controlled study in bogs in Germany[3] found that burning increased moss/lichen/bare ground cover in the short term (2–7 months after burning). Three replicated, paired studies (based on the same experimental set-up) in one bog in the UK[1,2,6] found that moss cover (including *Sphagnum*) was higher in plots burned more often.
- **Herb cover (4 studies):** Of two replicated, paired, controlled studies in bogs in Germany[3] and the UK[6], one[3] found that burning had no effect on cottongrass cover after 2–7 months but the other[6] found that burning increased cottongrass cover after 8–18 years. Two replicated, paired studies in the same bog in the UK[1,2] reported that cottongrass cover was similar in plots burned every 10 or 20 years. The study in Germany[3] also found that burning reduced purple moor grass cover but had mixed effects, amongst sites, on cover of other grass-like plants and forbs.
- **Tree/shrub cover (5 studies):** Three replicated, paired studies in a bog in the UK[1,2,6] found that heather cover was lower in plots burned more often. One replicated, paired, controlled study in bogs in Germany[3] found that heather cover was lower in burned than unburned plots. Two replicated, controlled studies in the bogs in Germany[3] and fens in the USA[5] found that burning (sometimes[5] along with other interventions) did not affect cover of other shrubs.
- **Vegetation structure (1 study):** One replicated, paired, controlled study in a bog in the UK[4] found that plots burned more frequently contained more biomass of grass-like plants than plots burned less often, but contained less total vegetation, shrub and bryophyte biomass.
- **Overall plant richness/diversity (2 studies):** Two replicated, controlled studies (one also randomized and paired) in the fens in the USA[5] and a bog in the UK[6] found that burning reduced or limited plant species richness. In the USA, burning was carried out along with other interventions.

Background

Prescribed burns can be used to manage problematic plants, especially shrubs and grasses that may overgrow and outcompete other desirable peatland vegetation like mosses (Chapman & Rose 1991). By removing above-ground vegetation, fire can also be used to manage the physical vegetation structure. The effect of fire on problematic plants is often temporary: many can regrow from roots or stumps that remain.

Frequent fires are not a natural feature of peatlands, with the exception of some fens (Middleton *et al.* 2006). Bogs and tropical peat swamps naturally burn every few *centuries* (Lindsay *et al.* 2011; Page & Hooijer 2016).

CAUTION: There are many risks to management by prescribed burning. For example, it can be difficult to control the intensity, duration and area of a prescribed burn. Uncontrolled burns can damage seed banks, *Sphagnum* mosses and the peat itself. Prescribed burns may be best carried out in the winter, when the peat is cold and wet (sometimes frozen), to avoid setting the peat on fire and reduce the risk of fire spreading beyond the prescribed area. Also note that burning might produce apparently desirable changes in vegetation over the short term (e.g. less heather cover and increased herb cover) followed by a rapid return to a degraded state.

Related interventions: prescribed burning to manage plants as part of a traditional disturbance regime (Section 8.9); interventions to address the threat from excess wild fire (Sections 8.10, 8.11, 8.12, 8.13 and 15.1).

Chapman S.B. & Rose R.J. (1991) Changes in the vegetation at Coom Rigg Moss National Nature Reserve within the period 1958–86. *Journal of Applied Ecology*, 28, 140–153.

Lindsay R., Birnie R. & Clough J. (2011) *Burning*. IUCN UK Peatland Programme Briefing Note No. 8.

Middleton B., Holsten B. & van Diggelen R. (2006) Biodiversity management of fens and fen meadows by grazing, cutting and burning. *Applied Vegetation Science*, 9, 307–316.

Page S.E. & Hooijer A. (2016) In the line of fire: the tropical peatlands of South East Asia. *Philosophical Transactions of the Royal Society B*, 371, 20150176.

A replicated, randomized, paired study in 1954–1973 in a blanket bog in England, UK (1) found that repeatedly burned plots developed less heather *Calluna vulgaris* cover and greater cover of some mosses than once-burned plots, but that cover of cottongrasses *Eriophorum* spp. was similar under both treatments. After 18 years, twice-burned plots consistently had less heather cover (19–30%) than once-burned plots (67–82%). Twice-burned plots also had greater cover of rusty swan-neck moss *Campylopus flexuosus* in 3 of 4 comparisons (for which twice-burned: 38–46%; once-burned: 11–22%) and *Sphagnum* mosses in 2 of 4 comparisons (when sheep were excluded; twice-burned: 19%; once-burned: 5%). All plots had similar cover of six other moss species (see original paper) and cottongrasses (twice-burned: 8–67%; once-burned: 6–62%). Mixed responses to burning were reported for 26 other plant groups. In 1954, four areas (each containing six 1,000 m^2 plots) in a historically grazed bog were burned. Within each area, two random plots were burned again in 1965 whilst four plots were not burned again. Half of the plots were also fenced to exclude sheep. In 1972, vegetation cover was estimated by recording, in each plot, plants touching 250 randomly placed pins. This study was based on the same experimental set-up as (2), (4) and (6).

A replicated, paired study in 1954–1980 in a blanket bog in England, UK (2) reported that burned plots became dominated by cottongrasses *Eriophorum* spp. within 3–5 years, and that burning more often reduced cover of heather *Calluna vulgaris* but typically increased moss and liverwort cover. These results were not tested for statistical significance. All plots were burned in 1954 and 1975. In 1978–1980, the plots were dominated by cottongrass (sheathed cottongrass *Eriophorum vaginatum* cover: 46–84%; common cottongrass *Eriophorum angustifolium* cover: 3–28%). Half of the plots were also burned in 1965. In 1978–1980, these plots had less cover of heather (shoots: 6–26%; stems: 2–17%) than plots that were not burned in 1965 (shoots: 22–70%; stems: 17–59%). The more frequently burned plots also had greater moss cover (*Sphagnum* in 5 of 6 comparisons; other mosses in 14 of 18 comparisons) and liverwort cover (in 22 of 33 comparisons) but similar cottongrass cover. In 1954, four pairs of 1,000 m^2 plots were established on a grazed bog. All plots were burned in 1954 and 1975. Four plots (one plot/pair) were also burned in 1965. Vegetation cover was estimated in August 1978–1980, in 128 quadrats/plot, (each 10 x 10 cm and arranged along a transect). This study was based on the same experimental set-up as (1), (4) and (6).

A replicated, paired, controlled study in 2003 in four degraded raised bogs in Germany (3) found that burned plots consistently had less cover than unburned plots of purple moor grass *Molinia caerulea* (25 vs 40%) and heather *Calluna vulgaris* (6 vs 20%), but more moss/lichen/bare ground cover (36 vs 8%). Meanwhile, burned and unburned plots had similar cover of cottongrass *Eriophorum vaginatum* (12 vs 16%) and cross-leaved heath *Erica tetralix* (2 vs 6%). Vegetation height, and cover of other minor forbs and grass-like plants, responded inconsistently to burning across the four bogs (see original paper). Between February and May 2003, 4–45 ha of four grassy/shrubby bogs were burned. Two fires were prescribed and two were wild. The study does not analyze the effects of these separately. Between July and September 2003, vegetation cover and height were recorded along a 100 m transect in each bog, spanning the burned area and an adjacent unburned area.

A replicated, paired, controlled study in 1954–2004 in a blanket bog in England, UK (4) found that repeatedly burned plots contained less total vegetation, shrub and bryophyte biomass than once-burned more biomass of grass-like plants. After 50 years, repeatedly burned plots contained less above-ground vegetation biomass (134 g/m^2) than once-burned plots (297 g/m^2). This included less biomass of shrubs (repeatedly burned: 236; once-burned: 116 g/m^2) and bryophytes (repeatedly burned: 5; once-burned: 53 g/m^2). In contrast, biomass of grass-like plants was significantly higher in repeatedly burned plots (13 g/m^2) than once-burned plots (8 g/m^2). In 1954, sixteen 1,000 m^2 plots were established, in four blocks of four, in a historically grazed bog. All plots were burned once in 1954. Thereafter, eight plots (two plots/block) were burned every 10 years. The other plots were not burned again. Additionally, half of the plots were fenced to exclude sheep. In 2003–2004, live above-ground vegetation was cut from one 25 cm^2 quadrat/plot, then dried and weighed. Samples were taken in spring, summer, autumn and winter. This study was based on the same experimental set-up as (1), (2) and (6).

A replicated, controlled, before-and-after study in 1986–2000 in two shrub-invaded fens in Ohio, USA (5) found that burning shrubs (along with cutting and herbicide application) altered plant community composition and prevented increases in plant species richness, but had no effect on shrub cover. The overall plant community composition changed significantly over time along transects with and without shrub control, but they accumulated different sets of species (data reported as a graphical analysis). Plant species richness was stable in the fen with shrub control (before: 12.8; after 14 years: 12.7 species/m^2) but increased in the fen without shrub control (before: 12.5; after 14 years: 14.6 species/m^2). Woody plant cover increased similarly in fens with shrub control (before: 46%; after 11 years: 62%) and without shrub control (before: 20%; after: 28%). From 1986, encroaching shrubs were managed using *ad hoc* burning, cutting and herbicide application. The study does not distinguish between the effects of these interventions. Three of four transects were managed in one fen ('with shrub control'). Only one of four transects were managed in the other fen ('without shrub control'). In summer 1986 (before shrub control began), 1999 and 2000, vegetation cover was estimated in 1 m^2 quadrats along the eight transects. Shrub cover was estimated from aerial photographs.

A replicated, randomized, paired, controlled study in 1954–2001 in a blanket bog in England, UK (6) found that repeated burning prevented development of heather-dominated vegetation and increased *Sphagnum* moss cover (with a short time between burns) but reduced total plant species richness. At first measurement in 1972, all plots had similar liverwort-rich vegetation. After 29 years, burned plots had greater cover of cottongrasses *Eriophorum* spp. than unburned plots, but less cover of heather *Calluna vulgaris* (data reported as graphical analyses). Averaged over the entire experimental period, plots burned every 10 years (but not plots burned every 20 years) had greater cover of *Sphagnum* moss than unburned plots (data not reported). There were significantly fewer plant species in burned plots (15.5–16.6/plot) than in unburned plots (17.3/plot). The effects of burning were similar in grazed and ungrazed plots. In 1954–1955, four 60 x 90 m areas were burned in a historically grazed bog. Within each area, two random plots were left unburned for the rest of the study period, two plots were burned every 10 years, and two plots burned every 20 years. Under each treatment, half of the plots were grazed by sheep. Vegetation cover was measured in 1972, 1982, 1991 and 2001 by recording, in each plot, plants touching 100 randomly placed pins. This study was based on the same experimental set-up as (1), (2) and (4).

(1) Rawes M. & Hobbs R. (1979) Management of semi-natural blanket bog in the northern Pennines. *Journal of Ecology*, 67, 789–807.
(2) Hobbs R.J. (1984) Length of burning rotation and community composition in high-level *Calluna-Eriophorum* bog in N England. *Vegetatio*, 57, 129–136.
(3) Hochkirch, A. & Adorf, F. (2007) Effects of prescribed burning and wildfires on Orthoptera in Central European peat bogs. *Environmental Conservation*, 34, 225–235.
(4) Ward S.E., Bardgett R.D., McNamara N.P., Adamson J.K. & Ostle N.J. (2007) Long-term consequences of grazing and burning on northern peatland carbon dynamics. *Ecosystems*, 10, 1069–1083.
(5) Barry M.J., Barbara A.K. & De Szalay F. (2008) Long-term plant community changes in managed fens in Ohio, USA. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 18, 392–407.
(6) Lee H., Alday J.G., Rose R.J., O'Reilly J. & Marrs R.H. (2013) Long-term effects of rotational prescribed burning and low-intensity sheep grazing on blanket-bog plant communities. *Journal of Applied Ecology*, 50, 625–635.

9.9 Use covers/barriers to control problematic plants

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- We captured no evidence for the effect on peatland vegetation of using covers or barriers to control problematic plants.

Background

Covers such as plastic sheeting or straw mulch could be used to control problematic plants. These may act as direct physical barriers (e.g. temporarily covering the peatland surface, to prevent seeds from establishing whilst they are viable) and/or indirect barriers that modify environmental conditions (e.g. opaque covers that block light, prevent photosynthesis and kill problematic plants). CAUTION: Covers may also kill desirable species. Temporary application, when the peatland is most vulnerable to invasion by problematic plants, could solve this problem.

9.10 Use herbicide to control problematic plants

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- **One study** examined the effect on peatland vegetation of using herbicide to control problematic plants. The study was in fens.
- **Plant community composition (1 study):** One replicated, controlled, before-and-after study in fens in the USA[1] found that applying herbicide to shrubs (along with other interventions) changed the overall plant community composition.
- **Tree/shrub cover (1 study):** The same study[1] found that applying herbicide to shrubs (along with other interventions) could not prevent increases in shrub cover over time.
- **Overall plant richness/diversity (1 study):** The same study[1] found that applying herbicide to shrubs (along with other interventions) *prevented* increases in plant species richness.

Background

Herbicides are plant-killing chemicals. They can be applied to an entire peatland area, or specifically applied to individual plants (typically shrubs and trees).

CAUTION: Herbicides are not recommended as general conservation tool. Many are not specific to the target plant so can cause collateral damage. Many have negative effects side effects on biodiversity, the environment and human health (Pimentel *et al.* 1992). Accordingly, herbicide use is being reduced or banned in many countries.

Pimentel D., Acquay H., Biltonen M., Rice P., Silva M., Nelson J., Lipner V., Giordano S., Horowitz A. & D'Amore M. (2006) Environmental and economic costs of pesticide use. *BioScience*, 42, 750–760.

A replicated, controlled, before-and-after study in 1986–2000 in two shrub-invaded fens in Ohio, USA (1) found that applying herbicide to shrubs (along with burning and cutting) altered plant community composition and prevented increases in plant species richness, but had no effect on shrub cover. The overall plant community composition changed significantly over time along transects with and without shrub control, but they accumulated different sets of species (data reported as a graphical analysis). Plant species richness was stable in the fen with shrub control (before: 12.8; after 14 years: 12.7 species/m^2) but increased in the fen without shrub control (before: 12.5; after 14 years: 14.6 species/m^2). Woody plant cover increased similarly in fens with shrub control (before: 46%; after 11 years: 62%) and without shrub control (before: 20%; after: 28%). From 1986, encroaching shrubs were managed using *ad hoc* herbicide application, burning and cutting. The study does not distinguish between the effects of these interventions. Three of four transects were managed in one fen ('with shrub control'). Only one of four transects were managed in the other fen ('without shrub control'). In summer 1986 (before shrub control began), 1999 and 2000, vegetation cover was estimated in 1 m^2 quadrats along the eight transects. Shrub cover was estimated from aerial photographs.

(1) Barry M.J., Barbara A.K. & De Szalay F. (2008) Long-term plant community changes in managed fens in Ohio, USA. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 18, 392–407.

9.11 Introduce an organism to control problematic plants Ⓑ Ⓕ Ⓢ

- **One study** examined the effect on peatland vegetation of introducing an organism (other than large vertebrate grazers) to control problematic plants. The study was in a fen meadow.
- **Plant community composition (1 study):** One controlled, before-and-after study in a fen meadow in Belgium[1] found that introducing a parasitic plant altered the overall plant community composition.
- **Vegetation cover (1 study):** The same study[1] found that introducing a parasitic plant reduced cover of the dominant sedge but increased moss cover.
- **Overall plant richness/diversity (1 study):** The same study[1] found that introducing a parasitic plant increased overall plant species richness.

Background

This section considers biological control: controlling problematic organisms by managing their enemies. Typically, this involves releasing natural enemies of target problematic organisms, such as microorganisms (e.g. a virus or a fungus), insects or parasitic plants. More enemies usually equals more damage to the target organisms. Biological control could be particularly effective for non-native problematic plants: their success in their new range may be due to escape from natural enemies in their native range (Keane & Crawley 2002).

CAUTION: Organisms introduced for biological control can themselves become problematic pests (e.g. the harlequin ladybird; Roy *et al.* 2016) or could damage non-target plants. Introductions should not be carried out without thorough assessment of likely negative impacts, non-target effects and effectiveness of control.

Related intervention: 'introduce' large grazing vertebrates, such as sheep or cows, to control problematic plants (Section 9.7).

Keane R.M. & Crawley M.J. (2002) Exotic plant invasions and the enemy release hypothesis. *Trends in Ecology and Evolution*, 17, 164–170.

Roy H.E., Brown P.M.J., Adriaens T. *et al.* (2016) The harlequin ladybird, *Harmonia axyridis*: global perspectives on invasion history and ecology. *Biological Invasions*, 18, 997–1044.

A controlled, before-and-after study in 1994–2012 in a degraded fen meadow in Belgium (1) found that a plot sown with parasitic marsh lousewort *Pedicularis palustris* developed a different plant community to unsown plots, less dominated by acute sedge *Carex acuta*, and with greater moss cover and more plant species. After six years, plots with and without lousewort contained a significantly different overall plant community (reported as a statistical model result). The plot with lousewort contained less acute sedge than the plot without lousewort: less biomass (80 vs 540 g/m^2), shorter plants (100 vs 40 cm) and less cover (after 18 years; 20 vs 80%). After six years, the plot with lousewort also contained less overall plant biomass (460 vs 670 g/m^2), but greater moss cover (49 vs 6%) and more plant species (21 vs 14 species/400 m^2). Before intervention, vegetation was similar in both plots (acute sedge biomass: 870 g/m^2; acute sedge cover: 100%; overall plant biomass: 960 g/m^2; other data not reported). In July 1994, one 20 x 20 m plot dominated by acute sedge was sown with 500 lousewort seeds. An adjacent plot was not sown and lousewort plants were continually removed. Biannual mowing had been resumed in 1992. Data were recorded in each plot in 1994, 2000 and 2012: height of 30 random sedge plants, dry above-ground biomass from six 1 m^2 quadrats, and plant species and moss cover in ten 1 m^2 quadrats.

(1) Decleer K., Bonte D. & van Diggelen R. (2013) The hemiparasite *Pedicularis palustris*: 'ecosystem engineer' for fen-meadow restoration. *Journal for Nature Conservation*, 21, 65–71.

Interventions: Problematic animals

9.12 Exclude wild herbivores using physical barriers

Ⓑ Ⓕ Ⓢ

- **One study** examined the effect on peatland vegetation of physically excluding wild herbivores. The study was in a fen meadow.
- **Vegetation cover (1 study):** One replicated, paired, controlled study in a fen meadow in Poland[1] reported that the effect of boar- and deer exclusion on vascular plant and moss cover depended on other treatments applied to plots.

- **Vegetation structure (1 study):** The same study[1] reported that the effect of boar- and deer exclusion on total vegetation biomass depended on other treatments applied to plots.
- **Overall plant richness/diversity (1 study):** The same study[1] reported that the effect of boar- and deer exclusion on plant species richness depended on other treatments applied to plots.

Background

Herbivores are animals that eat plants. Wild herbivores on temperate peatlands include deer, rabbits, hares, kangaroos, feral horses, feral pigs, grouse and slugs. Insects, monkeys and other large mammals are important herbivores in tropical peat swamps. Herbivores can damage peatland vegetation directly, by eating it. Herbivores can also have indirect effects on peatland vegetation. Large animals can trample and compact peat. Beavers, introduced to Tierra del Fuego, can flood existing peatlands when they build dams or drain peatlands through channels formed when dams fail (Grootjans et al. 2014). Wild herbivores could be physically excluded from pristine peatlands to prevent damage, or from damaged peatlands to let them recover.

Related interventions: exclude or remove domestic livestock, which may be the dominant herbivores on peatlands (Sections 3.5 and 3.6); use fences or barriers specifically to protect planted/sown peatland plants (Section 13.15).

Grootjans A., Iturraspe R., Fritz C., Moen A. & Joosten H. (2014) Mires and mire types of Peninsula Mitre, Tierra del Fuego, Argentina. *Mires and Peat*, 14, Article 1.

A replicated, paired, controlled study in 2004–2007 in a degraded fen meadow in Poland (1) found that the effect of fencing (to exclude wild herbivores) on vegetation depended on other treatments applied to plots: hay addition and topsoil stripping. This was true for plant species richness, vascular plant cover, moss cover and vegetation biomass (reported as statistical model results). For example, amongst areas stripped of 20 cm of topsoil, fencing increased plant species richness if hay was not added, but reduced richness if hay was added. These comparisons were not tested for statistical significance. In 2004, eight pairs of plots (8 x 16 m) were established in a drained fen meadow grazed by wild boar and deer. Eight plots (one plot/pair) were fenced to exclude these herbivores. The other plots were not fenced. Additionally, all plots were stripped of topsoil (20 or 40 cm deep), and parts of each plot were sown with hay from a nearby fen meadow (details not clear). Vegetation cover and plant species were recorded annually between 2004 (after stripping and fencing) and 2007. Total vegetation biomass was measured from clippings taken in August 2006–2007.

(1) Klimkowska A., Kotowski W., van Diggelen R., Grootjans A.P., Dzierża P. & Brzezińska K. (2010) Vegetation re-development after fen meadow restoration by topsoil removal and hay transfer. *Restoration Ecology*, 18, 924–933.

9.13 Control populations of wild herbivores

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of controlling populations of wild herbivores.

Background

Herbivores are animals that eat plants. Wild herbivores on temperate peatlands include deer, rabbits, hares, kangaroos, feral horses, feral pigs, grouse and slugs.

Insects, monkeys and other large mammals are important herbivores in tropical peatlands. Herbivores can damage peatland vegetation directly, by eating it. Herbivores can also have indirect effects on peatland vegetation. Large animals can trample and compact peat. Beavers, introduced to Tierra del Fuego, can flood existing peatlands when they build dams or drain peatlands through channels formed when dams fail (Grootjans et al. 2014). Controlling herbivore populations (e.g. by trapping, shooting or applying pesticides) could reduce these impacts.

CAUTION: These actions might have negative side effects for the rest of the food chain (e.g. less food for predators of the controlled animals, accumulation of poisons in non-target animals) or could directly kill non-target animals.

Related interventions: interventions to address the threat from domestic livestock, which may be the dominant herbivores on peatlands (Sections 3.5–3.9).

Grootjans A., Iturraspe R., Fritz C., Moen A. & Joosten H. (2014) Mires and mire types of Peninsula Mitre, Tierra del Fuego, Argentina. *Mires and Peat*, 14, Article 1.

10. Threat: Pollution

Background

Peatlands are vulnerable to a wide variety of pollutants from agriculture, residential areas, industry, vehicles, roads (e.g. storm water runoff and road salt), mining (e.g. heavy metals, sediments), fossil fuel extraction (e.g. fracking) and fossil fuel transport (e.g. oil spills).

In addition to water (see Chapter 8), peatlands and their vegetation are strongly defined by nutrient availability and pH or acidity (Rydin & Jeglum 2013). Peatlands tend to have low nutrient levels, although this varies within and between sites. Bogs are more acidic (have a low pH) than fens, and poor fens are more acidic than rich fens. Inputs of excess nutrients or water of the wrong pH can change the physical environment of peatlands, affecting the plants that grow within them (Bobbink *et al.* 1998; Lamers *et al.* 2002). Additionally, peatland plants can be killed by other generally harmful pollutants like herbicides, heavy metals and oil spills.

Polluted water can enter peatlands through surface or underground flows. Heavy metals, sulphur dioxides and nitrous oxides can enter peatlands from the air or in rainfall: a particular problem near/downwind of urban areas (Moors for the Future 2006).

This synopsis only includes studies that measured the responses of peatland vegetation to pollution control, not studies that only measured physical or chemical changes.

Related threats: drainage and flooding, which can cause changes in nutrient availability and acidity (Chapter 8); problematic plants, which may grow in polluted peatlands (Chapter 9). Related interventions: cutting/mowing herbaceous plants or removing plant litter, which may remove excess nutrients from peatlands (Chapter 8); removing/burying topsoil, which may be polluted (Chapter 12).

Bobbink R., Hornung M. & Roelofs J.G.M. (1998) The effects of air-borne nitrogen pollutants on species diversity in natural and semi-natural European vegetation. *Journal of Ecology*, 86, 717–738.

Lamers L.P., Smolders A.J.P. & Roelofs J.G.M. (2002) The restoration of fens in the Netherlands. *Hydrobiologia*, 478, 107–130.

Moors for the Future (2006) *Air pollution in the Peak District.* Moors for the Future Research Note No. 9.

Rydin H. & Jeglum J.K. (2013) *The Biology of Peatlands, Second Edition*. Oxford University Press, Oxford.

Key messages

Multiple sources of pollution

10.1 *Clean waste water before it enters the environment* **1 study**

Characteristic plants: One study in the Netherlands found that cleaning water entering a floating fen, along with other interventions to reduce pollution, allowed cover of mosses characteristic of low nutrient levels to increase.

Vegetation structure: The same study found that after the input water began to be cleaned, along with other interventions to reduce pollution, vascular plant biomass decreased.

10.2 ***Divert/replace polluted water source(s)***	**3 studies**

Characteristic plants: One study in a fen in the Netherlands found that after a nutrient-enriched water source was replaced, along with other interventions to reduce pollution, cover of mosses characteristic of low nutrient levels increased.

Vegetation cover: Two studies in bogs in the UK and Japan reported that after polluting water sources were diverted, sometimes along with other interventions, *Sphagnum* moss cover increased. Both studies reported mixed effects on different species of herbs.

10.3 ***Slow down input water to allow more time for pollutants to be removed***	**1 study**

Characteristic plants: One before-and-after study in a floating fen in the Netherlands found that after input water was rerouted on a longer path, along with other interventions to reduce pollution, cover of mosses characteristic of low nutrient levels increased.

Vegetation structure: The same study found that after the input water was rerouted on a longer path, along with other interventions to reduce pollution, vascular plant biomass decreased.

10.4 ***Retain or create buffer zones between pollution sources and peatlands***	**0 studies**

We captured no evidence for the effect on peatland vegetation of retaining or creating buffer zones between pollution sources and peatlands.

10.5 ***Use artificial barriers to prevent pollution entering peatlands***	**0 studies**

We captured no evidence for the effect on peatland vegetation of using artificial barriers to prevent pollution entering peatlands.

10.6 ***Reduce fertilizer or herbicide use near peatlands***	**0 studies**

We captured no evidence for the effect on peatland vegetation of reducing fertilizer or herbicide use in adjacent areas.

10.7 ***Manage fertilizer or herbicide application near peatlands***	**0 studies**

We captured no evidence for the effect on peatland vegetation of managing fertilizer or herbicide use in adjacent areas.

Agricultural and aquacultural effluents

10.8 ***Convert to organic agriculture or aquaculture near peatlands***	**0 studies**

We captured no evidence for the effect on peatland vegetation of converting to organic agriculture or aquaculture near peatlands.

10.9 ***Limit the density of livestock on farmland near peatlands***	**0 studies**

We captured no evidence for the effect on peatland vegetation of limiting the density of livestock on farmland near peatlands.

10.10 ***Use biodegradable oil in farming machinery***	**0 studies**

We captured no evidence for the effect on peatland vegetation of using biodegradable oil in farming machinery.

Industrial and military effluents

10.11 ***Remove oil from contaminated peatlands***	**0 studies**

We captured no evidence for the effect on peatland vegetation of removing oil from contaminated peatlands.

Airborne pollutants

10.12 *Remove pollutants from waste gases before they enter the environment* **1 study**

Plant richness/diversity: One study in bogs in Estonia reported that after dust filters were installed in industrial plants, along with a general reduction in emissions, the number of *Sphagnum* moss species increased but the total number of plant species decreased.

10.13 *Add lime to reduce acidity and/or increase fertility* **1 study**

Vegetation structure: One replicated, controlled study in a fen meadow in the Netherlands found that liming increased overall vegetation biomass (mostly grass).

10.14 *Drain/replace acidic water* **2 studies**

Vegetation cover: Two controlled studies in fens in the Netherlands reported that draining acidic water had mixed effects on cover of *Sphagnum* moss and herbs after 4–5 years, depending on the species and whether moss was also removed.

Overall plant richness/diversity: One controlled, before-and-after study in a fen in the Netherlands reported that draining and replacing acidic water increased plant species richness.

Interventions: Multiple sources of pollution

10.1 Clean waste water before it enters the environment

Ⓑ Ⓕ Ⓢ

- **One study** examined the effect, on peatland vegetation, of cleaning waste water before it enters the environment. The study was in a fen.
- **Characteristic plants (1 study):** One study in a floating fen in the Netherlands[1] found that after input water began to be cleaned (along with other interventions to reduce pollution), cover of mosses characteristic of low nutrient levels increased.
- **Vegetation structure (1 study):** The same study found that after input water began to be cleaned (along with other interventions to reduce pollution), vascular plant biomass decreased.

Background

Waste water could be cleaned before it is released into peatlands or before it enters the environment in general (ultimately reaching peatlands). Excess nutrients, salts, heavy metals, radioactive materials and organic compounds should be removed. Acidity (pH) should be adjusted. Hot water should be cooled. Waste water could be treated using traditional industrial methods, or in 'constructed wetlands' that contain plants and microorganisms to absorb or break down pollutants (Kadlec *et al.* 2000).

CAUTION: Peatland vegetation is very sensitive to water chemistry (Rydin & Jeglum 2013). If cleaned waste water is being discharged directly into a peatland, its chemistry should be carefully controlled to match natural input water. Note that bogs receive their water only as rain, so are likely to be harmed by any other water inputs.

Related intervention: remove atmospheric pollutants from waste gases before they enter the environment (Section 10.12).

Kadlec R.H., Knight R.L., Vymazal J., Brix H., Cooper P. & Haberl R. (2000) *Constructed Wetlands for Pollution Control: Processes, Performance, Design and Operation*. IWA Publishing, London.

Rydin H. & Jeglum J.K. (2013) *The Biology of Peatlands, Second Edition*. Oxford University Press, Oxford.

A study in 1984–2013 in a floating rich fen in the Netherlands (1) found that after installing water purification facilities (along with other interventions to reduce pollution), moss cover changed to species characteristic of lower nutrient levels whilst vascular plant biomass decreased. Over 25 years following intervention, four of seven moss species characteristic of low nutrient levels increased in cover (from 1–62% to 11–83%). Meanwhile, six of seven moss species characteristic of high nutrient levels decreased in cover (from 7–78% to 1–32%). Over 28 years, vascular plant biomass decreased from 1,123 g/m^2 to 287 g/m^2. Since the 1970s, water purification facilities were built to treat the fen water source (no further details reported), the water source was changed from a nutrient-rich river to a nutrient-poor lake, and the water was rerouted to allow more time for nutrient reduction. The study does not distinguish between the effects of these interventions. In addition, there was a general reduction in nutrient input from urban areas. In 1988 and 2013, cover of every moss species was recorded in a 25 x 200 m area. In 1984 and 2012, above-ground vascular plant biomass was collected, dried and weighed.

(1) Kooijman A.M., Cusell C., Mettrop I.S. & Lamers L.P.M. (2016) Recovery of target bryophytes in floating rich fens after 25 yr of inundation by base-rich surface water with lower nutrient contents. *Applied Vegetation Science*, 19, 53–65.

10.2 Divert/replace polluted water source(s) Ⓑ Ⓕ Ⓢ

- **Three studies** examined the effect, on peatland vegetation, of diverting or replacing polluted water source(s). Two studies were in bogs[1,2] and one was in a fen[3].
- **Characteristic plants (1 study):** One study in a fen in the Netherlands[3] found that after a nutrient-enriched water source was replaced (along with other interventions to reduce pollution), cover of mosses characteristic of low nutrient levels increased.
- **Vegetation cover (2 studies):** Two studies (one before-and-after) in bogs in the UK[1] and Japan[2] reported that after polluting water sources were diverted (sometimes[1] along with other interventions), *Sphagnum* moss cover increased. Both studies reported mixed effects on herb cover, depending on species.

Background

This section considers interventions that prevent polluted water from entering peatlands. These usually involve construction of new pipes, channels or waterways to divert polluted water away from a focal peatland. Clearly, this could create a pollution problem for another habitat – unless the polluted water is diverted into a waste water treatment system, such as a constructed wetland.

CAUTION: Peatland vegetation is very sensitive to water quantity and quality (Rydin & Jeglum 2013). If a polluted water source is removed from a fen, it may need to be replaced with an alternative clean water source to avoid the peatland drying out (e.g. Kooijman *et al.* 2016). The chemistry of this water should be carefully monitored or controlled to avoid changing the chemistry of the peatland. Ground water should generally not be added to bogs: they should receive water only as rain.

Related interventions: drain/replace acidic water, which focuses on the removal/replacement of a specific type of polluted water (Section 10.14); legally protect peatlands, including policies or laws to prevent pollution (Section 14.1).

Kooijman A.M., Cusell C., Mettrop I.S. & Lamers L.P.M. (2016) Recovery of target bryophytes in floating rich fens after 25 yr of inundation by base-rich surface water with lower nutrient contents. *Applied Vegetation Science*, 19, 53–65.

Rydin H. & Jeglum J.K. (2013) *The Biology of Peatlands, Second Edition*. Oxford University Press, Oxford.

A before-and-after study in 1972–1987 in a historically mined raised bog in England, UK (1) reported that after diversion of polluted inflow (along with rewetting), cover of *Sphagnum* moss, white sedge *Carex curta* and cottongrasses *Eriophorum* spp. increased, but cover of purple moor grass *Molinia caerulea* decreased. No statistical tests were carried out. *Sphagnum* was found in 7% of quadrats before intervention but 27% after, white sedge in 0.0% before but 0.8% after, and cottongrasses in 1.1% before and 1.5–1.7% after. In contrast, purple moor grass *Molinia caerulea* occurred in 100% of quadrats before intervention but only 74% after. Eighteen other herb, shrub and tree species showed variable responses (see original paper). In 1974, polluted inflow from adjacent farms was diverted away from a bog, whilst the water outflow was blocked to raise the water table. The study does not distinguish between the effects of these interventions. Vegetation cover was recorded before (1972–1973) and after (1987) intervention, as presence/absence of species in 8,945 contiguous 4 m^2 quadrats covering the whole site.

A study in 1980–2006 in a floating bog in Japan (2) found that after removing polluting water sources (sewage and tap water), cover of *Sphagnum* moss increased. Cover of vascular plant species showed mixed responses. Between 1980 and 2006, the area of moss hummocks (containing blunt-leaved bog moss *Sphagnum palustre*) increased from 5,900 m^2 to 8,500 m^2. The area of moss mats (dominated by feathery bog moss *Sphagnum cuspidatum*) increased from 420 m^2 to 1,010 m^2. Of nine abundant vascular plant species, cover of three decreased (including sedge *Carex thunbergii* and bogbean *Menyanthes trifoliata*), cover of three increased (including swamp millet *Isachne globosa*) and cover of three did not change (including common reed *Phragmites australis*). Historically, the lake under the bog was polluted by sewage from a hospital, discharge/leakage of tap water from a purification plant and runoff from a road. Interventions to reduce pollution were (a) construction of a sewage system in the 1960s and (b) pumping to remove tap water leakage from 2003. Deliberate tap water discharge also stopped in the 1960s. Road runoff continued. Vegetation cover was extracted from maps made in 1980 and 2006.

A study in 1984–2013 in a floating rich fen in the Netherlands (3) found that after replacing a nutrient-rich water source with lower-nutrient water (along with other interventions to reduce pollution), moss cover changed to species characteristic of lower nutrient levels and vascular plant biomass decreased. Over 25 years following intervention, four of seven moss species characteristic of low nutrient levels increased in cover (from 1–62% to 11–83%). Meanwhile, six of seven moss species characteristic of high nutrient levels decreased in cover (from 7–78% to 1–32%). Over 28 years, vascular plant biomass decreased from 1,123 g/m^2 to 287 g/m^2. Since the 1970s, the fen water source was changed from a nutrient-rich river to a nutrient-poor lake, the input water was rerouted on a longer path to allow more time for nutrient reduction, and water purification facilities were built. The study does not distinguish between the effects of these interventions. In addition, there was a general reduction in nutrient input from urban areas. In 1988 and 2013, cover of every moss species was recorded in a 25 x 200 m area. In 1984 and 2012, above-ground vascular plant biomass was collected, dried and weighed.

(1) Meade, R. (1992) Some early changes following the rewetting of a vegetated cutover peatland surface at Danes Moss, Cheshire, UK, and their relevance to conservation management. *Biological Conservation*, 61, 31–40.
(2) Tsujino R., Fujita N., Katayama M., Kawase D., Matsui K., Seo A., Shimamura T., Takemon Y., Tsujimura N., Yumoto T. & Ushimaru A. (2010) Restoration of floating mat bog vegetation after eutrophication damages by improving water quality in a small pond. *Limnology*, 11, 289–297.
(3) Kooijman A.M., Cusell C., Mettrop I.S. & Lamers L.P.M. (2016) Recovery of target bryophytes in floating rich fens after 25 yr of inundation by base-rich surface water with lower nutrient contents. *Applied Vegetation Science*, 19, 53–65.

10.3 Slow down input water to allow more time for pollutants to be removed

B Ⓕ S

- **One study** examined the effect, on peatland vegetation, of slowing down input water to allow more time for pollutants to be removed. The study was in a fen.
- **Characteristic plants (1 study):** One before-and-after study in a floating fen in the Netherlands[1] found that after input water was rerouted on a longer path (along with other interventions to reduce pollution), cover of mosses characteristic of low nutrient levels increased.
- **Vegetation structure (1 study):** The same study found that after input water was rerouted on a longer path (along with other interventions to reduce pollution), vascular plant biomass decreased.

Background

Polluted water entering a peatland could be slowed down, allowing more time for natural breakdown or removal of pollutants before the water reaches the peatland. This could be facilitated by making input channels longer (e.g. Kooijman *et al.* 2016) or building a dam in the input channels (e.g. Bootsma *et al.* 2002). This intervention is mainly relevant to fens and fen meadows, which are fed by inputs of ground water. Bogs and tropical peat swamps are mainly fed by inputs of rainfall.

Related interventions: clean waste water before it enters the environment (Section 10.1); divert/replace source(s) of polluted input water (Section 10.2).

Bootsma M.C., van den Broek T., Barendregt A. & Beltman B. (2002) Rehabilitation of acidified floating fens by addition of buffered surface water. *Restoration Ecology*, 10, 112–121.

Kooijman A.M., Cusell C., Mettrop I.S. & Lamers L.P.M. (2016) Recovery of target bryophytes in floating rich fens after 25 yr of inundation by base-rich surface water with lower nutrient contents. *Applied Vegetation Science*, 19, 53–65.

A before-and-after study in 1984–2013 in a floating rich fen in the Netherlands (1) found that after rerouting input water on a longer path (along with other interventions to reduce pollution), moss cover changed to species characteristic of lower nutrient levels, whilst vascular plant biomass decreased. Four of seven moss species characteristic of low nutrient levels increased in cover (from 1–62% four years before ditch extension to 11–83% eleven years after). Meanwhile, six of seven moss species characteristic of high nutrient levels decreased in cover (from 7–78% to 1–32%). Vascular plant biomass decreased from 1,123 g/m^2 eight years before ditch extension to 287 g/m^2 ten years after. In 1992, water entering the fen was rerouted on a longer path to allow more time for nutrient removal. The study does not distinguish between the effects of this intervention and the long term effects of two other interventions carried out since the 1970s: use of water purification facilities and switching the water source from a nutrient-rich river to a nutrient-poor lake. In 1988 and 2013, cover of every moss species was recorded in a 25 x 200 m area. In 1984 and 2012, above-ground vascular plant biomass was collected, dried and weighed.

(1) Kooijman A.M., Cusell C., Mettrop I.S. & Lamers L.P.M. (2016) Recovery of target bryophytes in floating rich fens after 25 yr of inundation by base-rich surface water with lower nutrient contents. *Applied Vegetation Science*, 19, 53–65.

10.4 Retain or create buffer zones between pollution sources and peatlands

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of retaining or creating buffer zones between pollution sources and peatlands.

Background

Buffer zones can separate a peatland from a pollution source, preventing pollution from reaching the peatland or slowing it down to allow more time for pollutants to break down. Buffer zones of existing vegetation could be retained around development (e.g. by not building right up to the edge of a peatland), or vegetated buffer zones could be specifically created. They may be planted with plants that can absorb or break down pollutants (Kadlec *et al.* 2000). Buffer zones could be harvested to provide income to support peatland conservation (Wantzen *et al.* 2006).

To be included as evidence in this section, studies must have reported the effect of buffer zones on focal protected peatlands. Studies that report effects on vegetation within a buffer zone (e.g. Hynninen *et al*. 2011) are not included: this vegetation is sacrificed (exposed to pollution) to protect the focal peatland.

Related interventions: use artificial barriers to prevent pollution entering peatland (Section 10.5).

Hynninen A., Hamberg L., Nousiainen H., Korpela L. & Nieminen M. (2011) Vegetation composition dynamics in peatlands used as buffer areas in forested catchments in southern and central Finland. *Plant Ecology*, 212, 1803–1818.

Kadlec R.H., Knight R.L., Vymazal J., Brix H., Cooper P. & Haberl R. (2000) *Constructed Wetlands for Pollution Control: Processes, Performance, Design and Operation*. IWA Publishing, London.

Wantzen K.M., Siqueira A., da Cunha C.N. & de Sá M.d.F.P. (2006) Stream-valley systems of the Brazilian Cerrado: impact assessment and conservation scheme. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 16, 713–732.

10.5 Use artificial barriers to prevent pollution entering peatlands

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of using artificial barriers to prevent pollution entering peatlands.

Background

Artificial barriers such as sand bags, rocks, plastic curtains, absorbent matting or ditches could be used to separate peatlands from a pollution source. Barriers could prevent pollution from reaching the peatland entirely, or slow it down so it has more time to break down before reaching the peatland. These barriers are likely to be most effective as a short-term intervention to extreme pollution events e.g. oil or chemical spills (Zoltai & Kershaw 1995).

Related interventions: retain or create vegetated buffer zones between pollution source and peatland (Section 10.4).

Zoltai S.C. & Kershaw G.P. (1995) *Large volume oil spill on land surface: the Vozey oil field, Russia*. Proceedings of the 18th Arctic and Marine Oil Spill Program Technical Seminar, Environment Canada, Ottawa, 1177–1186.

10.6 Reduce fertilizer or herbicide use near peatlands

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of reducing fertilizer or herbicide use in adjacent areas.

Background

Fertilizers and herbicides can have negative effects on peatland vegetation if they spill over from agricultural or domestic land (Smolders *et al.* 2010). Herbicides can kill plants directly. Run off from fertilized land can carry excess nutrients (such as nitrogen and phosphorous) into peatlands, altering their naturally low nutrient levels. Simply applying less fertilizer or herbicide to agricultural land near peatlands could reduce the amount spilling over into peatlands. Ultimately, reduced chemical application could be driven by legislation, financial incentives and/or education.

Related interventions: other techniques to reduce fertilizer or herbicide runoff into peatlands, without necessarily reducing the total amount applied (Section 10.7). In practice, the interventions from Section 10.6 and 10.7 will often be used simultaneously.

Smolders A.J.P., Lucassen E.C.H.E.T., Bobbink R., Roelofs J.G.M. & Lamers L.P.M. (2010) How nitrate leaching from agricultural lands provokes phosphate eutrophication in groundwater fed wetlands: the sulphur bridge. *Biogeochemistry*, 98, 1–7.

10.7 Manage fertilizer or herbicide application near peatlands

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of managing fertilizer or herbicide use in adjacent areas.

Background

Fertilizers and herbicides can have negative effects on peatland vegetation if they spill over from agricultural or domestic land (Smolders *et al.* 2010). Herbicides can kill plants directly. Run off from fertilized land can carry excess nutrients (such as nitrogen and phosphorous) into peatlands, altering their naturally low nutrient levels.

Various techniques could be used to reduce spillover of these chemicals into peatlands (without reducing the overall amount applied, although this could also be beneficial; Section 10.6). Applying fertilizers when plants are actively growing means a greater proportion of the nutrients are taken up by the plants. Avoiding chemical application before heavy rain reduces the amount that is immediately washed away. Ploughing or harrowing parallel to slopes avoids creating channels that carry chemicals off agricultural land towards peatlands (or other habitats). Planting cover crops could protect bare ground, and the chemicals applied to it, from rainfall and reduce the amount washed away. Ultimately, better chemical management could be driven by legislation, financial incentives and/or education.

Related intervention: reduce the overall amount of fertilizer or herbicide use near peatlands, without other management of its application (Section 10.6). In practice, the interventions from Section 10.6 and 10.7 will often be used simultaneously.

Smolders A.J.P., Lucassen E.C.H.E.T., Bobbink R., Roelofs J.G.M. & Lamers L.P.M. (2010) How nitrate leaching from agricultural lands provokes phosphate eutrophication in groundwater fed wetlands: the sulphur bridge. *Biogeochemistry*, 98, 1–7.

Interventions: Agricultural and aquacultural effluents

10.8 Convert to organic agriculture or aquaculture near peatlands

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of converting to organic agriculture or aquaculture near peatlands.

Background

Organic farming avoids the use of synthetic fertilizers and pesticides, instead relying on crop rotation, locally adapted crops/livestock, biological pest control and natural on-site fertilizers such as manure (European Commission 2017). If these principles are applied on terrestrial farms or in aquaculture systems close to peatlands, spillover of pollutants into peatlands may be reduced.

This section considers the overall effect of organic vs conventional farming on peatland vegetation. Organic farming may benefit biodiversity in general (Dicks *et al.* 2013), as well as reducing spillover of chemicals on to peatlands. Sections 10.5 and 10.6 consider physical means to reduce chemical spillover.

Related interventions: retain or create buffer zones between pollution source (e.g. agricultural land) and peatland (Section 10.4); reduce amount of fertilizer or herbicide used near peatlands (Section 10.6); manage fertilizer or herbicide application near peatlands, such as when or how they are applied (Section 10.7).

Dicks L.V., Ashpole J.E., Dänhardt J., James K., Jönsson A., Randall N., Showler D.A., Smith R.K., Turpie S., Williams D. & Sutherland W.J. (2013) *Farmland Conservation: Evidence for the Effects of Interventions in Northern and Western Europe*. Pelagic Publishing, Exeter.

European Commission (2017) *What is Organic Farming?* Available at http://ec.europa.eu/agriculture/organic/organic-farming/what-is-organic-farming. Accessed 1 August 2017.

10.9 Limit the density of livestock on farmland near peatlands

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of limiting the density of livestock on farmland near peatlands.

Background

Waste products from livestock can run off agricultural land and flow into peatlands. This is a particular problem when land is compacted due to trampling by many animals. Excess nutrients in peatlands could reduce their ability to support their characteristic, and sometimes diverse, vegetation. Reducing the number of livestock on land near peatlands would reduce the amount of excrement and compaction, potentially reducing nutrient inputs to peatlands.

Related interventions: interventions to reduce the impact of livestock grazing directly on peatlands (Sections 3.5–3.9).

10.10 Use biodegradable oil in farming machinery

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of using biodegradable oil in farming machinery.

Background

Lubricating oil could spill or leak from machinery used on agricultural land, or to harvest plants from natural peatlands. Using biodegradable oil could reduce impacts from oil spillage, as it will not persist in the environment (Dubowski *et al.* 2013).

Dubowski A.P., Zembrowski K., Rakowicz A., Palowski T., Weymann S. & Wojnilowicz L. (2013) Developing new-generation machinery for vegetation management on protected wetlands in Poland. *Mires and Peat*, 13, Article 11.

Interventions: Industrial and military effluents

10.11 Remove oil from contaminated peatlands

Ⓑ Ⓕ S

- We captured no evidence for the effect on peatland vegetation of removing oil from contaminated peatlands.

Background

Peatlands may be affected by oil spills. North American peatlands are particularly vulnerable to oil spills from pipelines or wells: oil is extracted from large reserves in Alaska and Alberta, then transported long distances over peatlands. Spontaneous vegetation recovery following oil spills can be slow (Racine 1994). Interventions to remove oil, or increase microbial activity to break down oil more rapidly, include washing, burning, tillage, aeration and fertilization (Jorgenson & Joyce 1994; ExxonMobil 2008). Historical oil spills, such as those from wrecked military vehicles (Ardron 2013), may be harder or impossible to clean.

CAUTION: Interventions to clean up oil spills could kill any surviving vegetation and/or churn oil into the peat, hindering long term recovery in some situations.

Related interventions: use artificial barriers to prevent pollution entering peatlands, or spreading from an initial source of pollution (Section 10.5); habitat creation and restoration (Chapter 12).

Ardron P.A. (2013) Impacts of conflict and war on peatland landscapes. Pages 221–231 in: I.D. Rotherham & C. Handley (eds.) *War & Peat.* Wildtrack Publishing, Sheffield.

ExxonMobil (2008) *Oil Spill Response Field Manual*. ExxonMobil, USA.

Jorgenson M.T. & Joyce M.R. (1994) Six strategies for rehabilitating land disturbed by oil development in arctic Alaska. *Arctic*, 47, 374–390.

Racine C.H. (1994) Long-term recovery of vegetation on two experimental crude oil spills in interior Alaska black spruce taiga. *Canadian Journal of Botany*, 72, 1171–1177.

Interventions: Airborne pollutants

10.12 Remove pollutants from waste gases before they enter the environment

Ⓑ Ⓕ Ⓢ

- **One study** examined the effect on peatland vegetation of removing pollutants from waste gases before release into the environment. The study was in bogs.
- **Plant richness/diversity (1 study):** One before-and-after study in bogs in Estonia[1] reported that following installation of dust filters in industrial plants (along with a general reduction in emissions), the number of *Sphagnum* moss species increased but the total number of plant species decreased.

Background

Atmospheric pollutants can be removed from waste gases (e.g. from industry or transport) before they enter the environment. Physical or electrostatic filters can trap dust and ash particles. Sulphur dioxide can be removed by spraying alkaline substances (such as seawater) into waste gases. Reducing emissions of atmospheric pollutants may prevent damage to peatland vegetation or allow it to recover.

Related intervention: clean waste water before it is released (Section 10.1).

A before-and-after study in 1990–2007 in two raised bogs in Estonia (1) reported that after installing improved dust filters in industrial plants (along with a general reduction in emissions), total plant species richness decreased but *Sphagnum* moss species richness increased. These results were not tested for statistical significance. In the late 1980s/early 1990s, when bogs were polluted by calcium-rich ash, there were 91–123 plant species and nine *Sphagnum* species/0.1 ha. In 2007, after pollution was reduced, there were only 43–58 plant species but 14 *Sphagnum* species/0.1 ha. Throughout the 1990s, emissions of calcium-rich ash fell by 80%, partly through fitting improved dust filters but partly through reduced industrial activity. The study does not distinguish between the effects of these changes. In 2007, plant species were recorded in a 0.1 ha plot in each bog. Species richness was compared to published records from the late 1980s/early 1990s.

(1) Paal J., Vellak K., Liira J. & Karofeld E. (2009) Bog recovery in northeastern Estonia after the reduction of atmospheric pollution input. *Restoration Ecology*, 18, 387–400.

10.13 Add lime to reduce acidity and/or increase fertility Ⓑ Ⓕ Ⓢ

- **One study** examined the effect of liming (without planting) on peatland vegetation. The study was in a fen meadow. *N.B. Sections 12.1 and 13.1 consider liming in different contexts.*
- **Vegetation structure (1 study):** One replicated, controlled study in a fen meadow in the Netherlands[1] found that liming increased overall vegetation biomass (mostly grass).

Background

Peatland plant survival and growth is partly determined by the acidity of a peatland, or pH (Rydin & Jeglum 2013). Fen plants grow in alkaline to weakly acidic peat (approximately pH 6–8, similar to saliva, tap water or sea water). Bog plants grow in more acidic peat (approximately pH 4–5, similar to tomato juice or coffee). Lime (calcium and/or magnesium-rich chemicals) can be added if the peat becomes too acidic for a desired plant community. Liming can reduce acidity. It can also affect nutrient availability: nutrients such as phosphorous become locked away in acidic soils (Weil & Brady 2016).

CAUTION: The benefits and harms of liming are very context specific. Because fen plants require the least acidic conditions, liming is mostly used to manage fens and fen meadows. Liming may be useful in some bogs that have become extremely acidic (e.g. as a result of exceptionally acidic rain). In most bogs, liming could cause damage by removing natural acidity.

Related interventions: rewetting, which may reverse drainage-related acidification of surface peat (Section 8.1); restoring peatlands using multiple interventions, because lime is often used as part of a suite of interventions (Section 12.1); add lime to complement planting (Section 13.1).

Weil R.R. & Brady N.C. (2016) *The Nature and Properties of Soils, Fifteenth Edition*. Pearson, USA.
Rydin H. & Jeglum J.K. (2013) *The Biology of Peatlands, Second Edition*. Oxford University Press, Oxford.

A replicated, controlled study in 1994 in a degraded fen meadow in the Netherlands (1) found that limed plots contained more plant biomass after three months than unlimed plots. This was true in plots that had previously been stripped of topsoil (limed: 40; unlimed: 20 g/m² biomass) and plots that had not been stripped (limed: 250; unlimed: 200 g/m² biomass). The biomass was mostly established, dominant, velvety bentgrass *Agrostis canina* (precise contribution not reported). In May 1994, ten 1 m² plots in a degraded, historically drained fen meadow were limed (approximately 500 g/m²). Ten additional plots were not limed. Five limed and five unlimed plots had been previously stripped of topsoil. In August 1994, above-ground vegetation was harvested in one 60 x 60 cm quadrat/plot, then dried and weighed.

(1) van Duren I.C., Strykstra R.J., Grootjans A.P., ter Heerdt G.N.J. & Pegtel D.M. (1998) A multidisciplinary evaluation of restoration measures in a degraded *Cirsio-Molinietum* fen meadow. *Applied Vegetation Science*, 1, 115–130.

10.14 Drain/replace acidic water B Ⓕ S

- **Two studies** examined the effect on peatland vegetation of draining/replacing acidic surface water. Both studies were in fens.

- **Vegetation cover (2 studies):** Two controlled studies in fens in the Netherlands[1,2] reported that draining acidic water had mixed effects on cover of *Sphagnum* moss and herbs after 4–5 years, depending on the species and whether moss was also removed.
- **Overall plant richness/diversity (1 study):** One controlled, before-and-after study in a fen in the Netherlands[2] reported that draining and replacing acidic water increased plant species richness.

Background

Peatlands can hold onto acidic water, especially in thick carpets of moss. This is a natural process in the development of peatlands from fens to bogs (Rydin & Jeglum 2013). However, where the desired plant community is a fen, intervention may be needed to reduce acidity. Fen plants prefer weakly acidic, neutral or even alkaline peat (approximately pH 6–8, similar to saliva, tap water or sea water). Ditches can be dug to drain excess surface water (especially during periods of heavy rain). This may be replaced by less acidic water naturally, or with further intervention.

CAUTION: Peatland vegetation is very sensitive to water quantity and quality (Rydin & Jeglum 2013). If acidic water is drained, it will probably have to be replaced to prevent the peatland drying out. The chemistry of this water should be carefully monitored or controlled to avoid changing the chemical conditions of the peatland. This intervention is generally not applicable to bogs, which are naturally acidic.

Related interventions: remove plants, including mosses that can store acidic water (Section 9.2); divert/replace other sources of polluted water (Section 10.2).

Rydin H. & Jeglum J.K. (2013) *The Biology of Peatlands, Second Edition*. Oxford University Press, Oxford.

A controlled study in 1989–1993 in a degraded floating fen in the Netherlands (1) reported that draining acidic surface water increased herb cover, but only when moss was also removed. These results were not tested for statistical significance. Drainage alone had no effect on vegetation cover. After four years, a drained and undrained plot had similar vegetation cover (total moss: 93%; *Sphagnum*: 62%; herbs: 3%). However, drainage in combination with moss removal favoured herbs. A plot drained and cleared of moss developed 76% herb cover with 0% moss cover. In contrast, a plot cleared of moss but not drained regained high moss cover (total moss: 96%; *Sphagnum*: 53%; herbs: <1%). In 1989, a ditch was built to drain surface water from two 16 m^2 plots in an acidified, nutrient-enriched fen. Two other plots were not drained. In 1991, the moss carpet was also cleared from one drained and one undrained plot. In 1995, vegetation cover was recorded in six 1 m^2 quadrats/plot.

A controlled, before-and-after study in 1991–1996 in a degraded floating fen in the Netherlands (2) reported that draining acidic surface water (and replacing it with less acidic water) increased plant species richness and *Sphagnum* moss cover, but had no effect on sedge or common reed abundance. These results were not tested for statistical significance. Before intervention, plots contained approximately 16 species. After five years, drained plots contained 22–43 plant species, compared to 14–16 species in undrained plots. Drained plots had 41–100% *Sphagnum* cover, compared to 21–40% in undrained plots. Drain and undrained plots had similar cover of sedge *Carex nigra* (0–20%) and abundance of common reed *Phragmites australis* (in 81–100% of quadrats). Effects of drainage on cottongrass *Eriophorum angustifolium* abundance were more complicated and depended on whether moss was also removed (see original paper). In January 1992, ditches were built to drain surface water from one plot in an acidified, nutrient-enriched fen. An inflow of less acidic water was also created. Water was not manipulated in a neighbouring plot. Within each plot, surface moss was cleared from three subplots but not three others. Between 1991 and 1996, vegetation was estimated in quadrats covering 6–8 m^2 of each subplot.

(1) Beltman B., van den Broek T., Bloemen S. & Witsel C. (1996) Effects of restoration measures on nutrient availability in a formerly nutrient-poor floating fen after acidification and eutrophication. *Biological Conservation*, 78, 271–277.

(2) Bootsma M.C., van den Broek T., Barendregt A. & Beltman B. (2002) Rehabilitation of acidified floating fens by addition of buffered surface water. *Restoration Ecology*, 10, 112–121.

N.B. Results from (1) are also reported in: Beltman B., van den Broek T. & Bloemen S. (1995) Restoration of acidified rich-fen ecosystems in the Vechtplassen area: successes and failures. Pages 274–286 in: B.D. Wheeler, S.C. Shaw, W.J. Fojt & R.A. Robertson (eds.) *Restoration of Temperate Wetlands*. John Wiley & Sons Ltd., Chichester.

11. Threat: Climate change and severe weather

Background

This chapter addresses threats from long-term climatic change and extreme weather events. Peatlands are very sensitive to changes in rainfall, temperature, wind speed and salinity. Prolonged droughts, such as those associated with El Niño events in tropical peat swamps, could dry out peatlands and increase susceptibility to fire (Field *et al.* 2009). Intense rainfall could erode peat or flood peatlands. Rising sea levels and coastal storms could flood peatlands with salt water. The impacts of climate change are often worse in peatlands already degraded by other threats such as overgrazing, drainage and wild fire (Parish *et al.* 2008; Field *et al.* 2009).

There may be some capacity for peatlands to develop in new areas in response to climate change, but this may be limited as suitable areas shift towards the poles or upland peaks. It is likely that the land area suitable for some peatland types to form will shrink under climate change (Gallego-Sala & Prentice 2012).

Note that conservation of peatlands could in itself reduce the severity of climate change. Bogs alone are a key store of carbon, containing at least 20% of all carbon stored in terrestrial ecosystems, but covering only 2–3% of the global land surface (Heijmans *et al.* 2008). Healthy peatland vegetation contributes to effective carbon storage (e.g. Loisel & Yu 2013). Information on the effects of conservation interventions on carbon storage *per se* is beyond the scope of this synopsis.

Related threats: wild fire (Chapter 8 and Chapter 15); problematic plants, which may develop in peatlands affected by climate change (Chapter 9). Related interventions: general habitat creation and restoration (Chapter 12).

Field R.D., van der Werf G.R. & Shen S.S.P. (2009) Human amplification of drought induced biomass burning in Indonesia since 1960. *Nature Geoscience*, 2, 185–188.

Gallego-Sala A.V. & Prentice I.C. (2012) Blanket peat biome endangered by climate change. *Nature Climate Change*, 3, 152–155.

Heijmans M.M.P.D., Mauquoy D., van Geel B. & Berendse F. (1998) Long-term effects of climate change on vegetation and carbon dynamics in peat bogs. *Journal of Vegetation Science*, 19, 307–320.

Loisel J. & Yu Z. (2013) Surface vegetation patterning controls carbon accumulation in peatlands. *Geophysical Research Letters*, 40, 5508–5513.

Parish F., Sirin A., Charman D., Joosten H., Minayeva T., Silvius M. & Stringer L. (eds.) (2008) *Assessment on Peatlands, Biodiversity and Climate Change: Main Report*. Global Environment Centre, Kuala Lumpur and Wetlands International, Wageningen.

Key messages

11.1 Add water to peatlands to compensate for drought **0 studies**

We captured no evidence for the effect on peatland vegetation of adding water to peatlands to compensate for drought.

11.2 Plant shelter belts to protect peatlands from wind **0 studies**

We captured no evidence for the effect on peatland vegetation of planting shelter belts to protect peatlands from wind.

11.3 Build barriers to protect peatlands from the sea **0 studies**

We captured no evidence for the effect on peatland vegetation of building barriers to protect peatlands from seawater damage.

11.4 Restore/create peatlands in areas that will be climatically suitable in the future **0 studies**

We captured no evidence for the effect on peatland vegetation of restoring or creating peatlands in areas that will be climatically suitable in the future.

Interventions

11.1 Add water to peatlands to compensate for drought Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of adding water to peatlands to compensate for drought.

Background

Peatland vegetation can only grow in wet areas. The plants depend on the water itself, but also the chemistry of wet peat (if peat dries out, acidity and nutrient levels can change though chemical reactions; Lamers *et al.* 2002). As a short-term intervention to compensate for drought, water could be diverted to peatlands (e.g. from rivers; Roelofs 1991).

CAUTION: Peatland vegetation is very sensitive to water quality (Rydin & Jeglum 2013). To avoid altering the chemistry of the peatland, a suitable water source must be chosen: of the correct pH (not too acidic, not to alkaline) and with no unusual nutrients/chemicals. Note that bogs receive their water only as rain, so ground water addition should be avoided if possible.

Related interventions: rewetting, as a more long-term intervention (Section 8.1); interventions to address wild fires, which could be associated with droughts (Sections 8.10, 8.11, 8.12, 12.5 and 15.1).

Roelofs J.G.M. (1991) Inlet of alkaline river water into peaty lowlands: effects on water quality and *Stratiotes aloides* L. stands. *Aquatic Botany*, 39, 267–293.

Rydin H. & Jeglum J.K. (2013) *The Biology of Peatlands, Second Edition*. Oxford University Press, Oxford.

11.2 Plant shelter belts to protect peatlands from wind Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of planting shelter belts to protect peatlands from wind.

Background

Rows of trees can be planted as windbreaks, which could prevent excess drying and erosion of surface peat. Strong winds and storms may become more common with climate change. Some trees do not grow in wet peat but could be planted in natural or created mounds, or along the edges of other land uses such as roads or farms.

Related interventions: retain/create pollution buffer zones (Section 10.4); build fire breaks (Section 8.12). These could both be dual functions of shelter belts.

11.3 Build barriers to protect peatlands from the sea

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of building barriers to protect peatlands from seawater damage.

Background

Some peatlands are in low-lying and/or coastal areas (e.g. bogs in eastern Canada, fens in eastern England) and are therefore vulnerable to seawater flooding, either due to temporary storm surges or longer-term sea level rise. Peatlands with a lowered surface (as a result of peat extraction, compaction or subsidence) are especially vulnerable to flooding. Barriers could be built to separate the peatland from the sea.

If saltwater influx does occur, peatlands can be permanently damaged. Some peatland plant species, notably *Sphagnum* moss, cannot tolerate salinity (Ward 2013). Therefore, any restoration attempts might focus on introducing salt tolerant vegetation rather than attempting to restore the former peatland characteristic vegetation (e.g. Emond *et al.* 2016).

Related interventions: use artificial barriers to prevent pollution entering peatlands (Section 10.5); habitat creation and restoration (Chapter 12).

Emond C., Lapointe L., Hugron S. & Rochefort L. (2016) Reintroduction of salt marsh vegetation and phosphorus fertilisation improve plant colonisation on seawater-contaminated cutover bogs. *Mires and Peat*, 18, Article 17.

Ward A. (2013) *Salinity tolerance of four bryophyte species* Sphagnum palustre, Sphagnum subsecundum, Mnium hornum *and* Aulacomnium palustre*, living in a sea-level fen*. Masters thesis, Central Connecticut State University, USA.

11.4 Restore/create peatlands in areas that will be climatically suitable in the future

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland vegetation of restoring or creating peatlands in areas that will be climatically suitable in the future.

Background

It may be wise to prioritize restoration or creation of peatland habitats in areas that will remain or become climatically suitable in the future (rather than areas that are destined to become unsuitable). These areas will also provide space for peatland vegetation to move into as current peatlands become too dry/wet/warm to support their characteristic plants (Oliver *et al.* 2012).

Related interventions: restoring water levels and management processes (Chapter 8) and other habitat creation and restoration interventions (Chapter 12).

Oliver T.H., Smithers R.J., Bailey S., Walmsley C.A. & Watts K. (2012) A decision framework for considering climate change adaptation in biodiversity conservation planning. *Journal of Applied Ecology*, 49, 1247–1255.

12. Habitat creation and restoration

Background

This chapter addresses creation and restoration of peatlands. Interventions here could be used to address multiple threats from previous chapters (e.g. to restore a peatland after mining or agriculture or severe weather).

We define restoration as 'returning peatland vegetation from a disturbed or altered condition towards a previously existing condition' (Mitsch & Gosselink 2015). In this sense restoration may, but almost always does not, return the vegetation exactly to that previous condition. It may be impossible to return to a previous condition if the physical habitat has been permanently changed. Further, because ecosystems naturally change in character through time, the previous condition may not reflect how the peatland would be now had it not been disturbed (Hughes *et al.* 2012).

Some studies involve planting trees to restore peatland habitats. Trees are a natural feature of some peatlands (e.g. forested bogs/fens and tropical peat swamp forests). It may be desirable to replace them after disturbances such as fire, logging or construction. However, trees are a threat to naturally open or sparsely forested peatlands: hence the interventions elsewhere in the synopsis that remove or control trees (Sections 3.3, 3.4, 8.6 and 9.6).

Within this chapter, there are separate sections for (a) combinations of multiple interventions, where it is difficult to separate the effects of any single intervention (b) interventions that only modify the physical habitat, creating more favourable conditions for peatland plants but relying on natural colonization and (c) interventions that involve actively introducing vegetation to kick start development of a natural peatland, usually with some modification of the physical environment beforehand or afterwards to create suitable conditions.

Related interventions: interventions tackling specific threats (Chapters 2–11); interventions to complement planting of peatland vegetation (Chapter 13); encourage restoration with legislation or agreements to protect peatlands (Chapter 14).

Hughes F.M.R., Adams W.M. & Stroh P.A. (2012) When is open-endedness desirable in restoration projects? *Restoration Ecology*, 20, 291–295.

Mitsch W.J. & Gosselink J.G. (2015) *Wetlands, Fifth Edition*. Wiley, New Jersey.

Key messages

General habitat creation and restoration

12.1 *Restore/create peatland vegetation (multiple interventions)* **9 studies**

Plant community composition: One replicated, controlled, before-and-after study in the UK reported that the overall plant community composition differed between restored and unrestored bogs. One replicated, controlled, site comparison study in Estonia found that restored and natural bogs contained more similar plant communities than unrestored and natural bogs. However, one site comparison study in Canada reported that after five years, bogs being restored as fens contained a different plant community to natural fens.

Characteristic plants: One controlled study, in a fen in France, reported that restoration interventions increased cover of fen-characteristic plants.

Moss cover: Five studies (one replicated, paired, controlled, before-and-after) in bogs or other peatlands in the UK, Estonia and Canada found that restoration interventions increased total moss or bryophyte cover. Two studies (one replicated and controlled) in bogs in the Czech Republic and Estonia reported that restoration interventions increased *Sphagnum* moss cover, but one replicated before-and-after study in bogs in the UK reported no change in *Sphagnum* cover following intervention. Two site comparison studies in Canada reported that after 1–15 years, restored areas had lower moss cover than natural fens.

Herb cover: Five studies (one replicated, paired, controlled, before-and-after) in bogs or other peatlands in the Czech Republic, the UK, Estonia and Canada reported that restoration interventions increased cover of herbs, including cottongrass and other grass-like plants.

Overall vegetation cover: Three studies (one replicated, controlled, before-and-after) in bogs in the UK and France reported that restoration interventions increased overall vegetation cover.

12.2 *Restore/create peatland vegetation using the moss layer transfer technique* **4 studies**

Plant community composition: One replicated study in bogs in Canada reported that the majority of restored areas developed a community of bog-characteristic plant species within 11 years. One controlled, before-and-after study in a bog in Canada reported that a restored area (included in the previous study) developed a more peatland-characteristic plant community over time, and relative to an unrestored area.

Vegetation cover: Two controlled studies in one bog in Canada reported that after 4–8 years, a restored area had greater cover than an unrestored area of mosses and bryophytes (including *Sphagnum*) and herbs (including cottongrass), but less cover of shrubs. One of the studies reported that vegetation in the restored area became more similar to local natural bogs.

Overall plant richness/diversity: One controlled, before-and-after study in a bog in Canada reported that after eight years, a restored area contained more plant species than an unrestored area.

Modify physical habitat only

12.3 *Fill/block ditches to create conditions suitable for peatland plants* **3 studies**

Vegetation cover: Two studies, in a bog in the UK and a fen in the USA, reported that blocked or filled ditches were colonized by peatland vegetation within 2–3 years. In the USA, vegetation cover was restored to natural, undisturbed levels. One replicated study in bogs in the UK reported that plants had not colonized blocked gullies after six months.

Overall plant richness/diversity: One site comparison study in a fen in the USA found that a filled ditch contained more plant species than adjacent undisturbed fen, after two years.

12.4 *Excavate pools* **2 studies**

Plant community composition: One replicated, before-and-after, site comparison study in bogs in Canada reported that excavated pools were colonized by some peatland vegetation over 4–6 years, but contained different plant communities to natural pools. In particular, cattail was more common in created pools.

Vegetation cover: One replicated, before-and-after, site comparison study in bogs in Canada reported that after four years, created pools had less cover than natural pools of *Sphagnum* moss, herbs and shrubs.

Overall plant richness/diversity: One replicated, before-and-after, site comparison study in bogs in Canada reported that after six years, created pools contained a similar number of plant species to natural pools.

12.5 ***Reprofile/relandscape peatland***	**1 study**

Plant community composition: One site comparison study in Canada reported that after five years, reprofiled and rewetted bogs (being restored as fens) contained a different plant community to nearby natural fens.

Vegetation cover: The same study reported that after five years, reprofiled and rewetted bogs (being restored as fens) had lower vegetation cover than nearby natural fens (specifically *Sphagnum* moss, other moss and vascular plants).

12.6 ***Roughen peat surface to create microclimates***	**0 studies**

We captured no evidence for the effect of roughening the peat surface to create microclimates (without planting afterwards) on peatland vegetation.

12.7 ***Remove upper layer of peat/soil***	**10 studies**

Plant community composition: Five studies (one replicated, randomized, paired, controlled) in a peatland in the USA and fens or fen meadows in the Netherlands and Poland reported that plots stripped of topsoil developed different plant communities to those in unstripped peatlands. In one study, the effect of stripping was not separated from the effect of rewetting. Two studies in fen meadows in Germany and Poland reported that the depth of soil stripping affected plant community development.

Characteristic plants: Four studies (one replicated, randomized, paired, controlled) in fen meadows in Germany and the Netherlands, and a peatland in the USA, reported that stripping soil increased cover of wetland- or peatland-characteristic plants after 4–13 years. In the Netherlands, the effect of stripping was not separated from the effect of rewetting. One replicated site comparison study in fens in Belgium and the Netherlands found that stripping soil increased fen-characteristic plant richness.

Herb cover: Three studies (one replicated, paired, controlled) in fens or fen meadows in Germany, the UK and Poland found that stripping soil increased rush, reed or sedge cover after 2–6 years. One controlled study in a fen meadow in the Netherlands reported that stripping soil did not affect sedge or bentgrass cover after five years. Two controlled studies, in fens or fen meadows in the Netherlands and the UK, found that stripping soil reduced purple moor grass cover for 2–5 years.

Vegetation structure: Two studies, in fens or fen meadows in the Netherlands and Belgium, found that stripping soil reduced vegetation biomass (total or herbs) for up to 18 years. One replicated, randomized, paired, controlled study in a peatland in the USA found that stripping soil had no effect on vegetation biomass after four years.

Overall plant richness/diversity: Three studies (one replicated, paired, controlled) in fens or fen meadows in the UK, Belgium and the Netherlands reported that stripping soil increased total plant species richness over 2–18 years. In one study, the effect of stripping was not separated from the effect of rewetting. One replicated, controlled study in a fen in Poland found that stripping soil had no effect on plant species richness after three years. One replicated, randomized, paired, controlled study in a peatland in the USA found that stripping soil increased plant species richness and diversity, after four years, in one field but decreased it in another. One replicated study in a fen meadow in Poland reported that plant species richness increased after soil was stripped.

12.8 ***Bury upper layer of peat/soil***	**0 studies**

We captured no evidence for the effect of burying the upper layer of peat or soil (without planting afterwards) on peatland vegetation.

12.9 ***Disturb peatland surface to encourage growth of desirable plants***	**2 studies**

Plant community composition: Two replicated, paired, controlled, before-and-after studies (one also randomized) in fens in Germany and Sweden reported that soil disturbance affected development of the plant community over 2–3 years. In Germany, disturbed plots developed greater cover of weedy species from the seed bank than undisturbed plots. In Sweden, the community in disturbed and undisturbed plots became less similar over time.

Characteristic plants: The same two studies reported that wetland- or fen-characteristic plants colonized plots that had been disturbed (along with other interventions). The study in Germany noted that no peat-forming species colonized the fen.

12.10 *Add inorganic fertilizer* **3 studies**

Vegetation cover: One replicated, randomized, paired, controlled, before-and-after study in a bog in New Zealand reported that fertilizing typically increased total vegetation cover.

Vegetation structure: One replicated, paired, controlled study in a fen meadow in the Netherlands found that fertilizing with phosphorous typically increased total above-ground vegetation biomass, but other chemicals typically had no effect.

Overall plant richness/diversity: One replicated, randomized, paired, controlled, before-and-after study in a bog in New Zealand reported that fertilizing typically increased plant species richness.

Growth: One replicated, controlled, before-and-after study in a bog in Germany found that fertilizing with phosphorous typically increased herb and shrub growth rate, but other chemicals had no effect.

Other: Three replicated, controlled studies in a fen meadow in Germany and bogs in Germany and New Zealand reported that effects of fertilizer on peatland vegetation were more common when phosphorous was added, than when nitrogen or potassium were added.

12.11 *Cover peatland with organic mulch* **2 studies**

Vegetation cover: One replicated, randomized, paired, controlled, before-and-after study in a bog (being restored as a fen) in Canada found that mulching bare peat did not affect cover of fen-characteristic plants. One replicated, controlled, before-and-after study in a bog in Australia reported that plots mulched with straw had similar *Sphagnum* moss cover to unmulched plots.

Characteristic plants: One replicated, randomized, paired, controlled, before-and-after study in a bog (being restored as a fen) in Canada found that covering bare peat with straw mulch increased the number of fen characteristic plants present, but did not affect their cover.

12.12 *Cover peatland with something other than mulch* **2 studies**

Vegetation cover: One replicated, controlled, before-and-after study in a bog in Germany reported that covering bare peat with fleece or fibre mats did not affect the number of seedlings of five herb/shrub species. One replicated, controlled, before-and-after study in bogs in Australia reported that recently-burned plots shaded with plastic mesh developed greater cover of native plants, forbs and *Sphagnum* moss than unshaded plots.

12.13 *Stabilize peatland surface to help plants colonize* **1 study**

Vegetation cover: One controlled, before-and-after study in a bog in the UK found that pegging coconut fibre rolls onto almost-bare peat did not affect the development of vegetation cover (total, mosses, shrubs or cottongrasses).

12.14 *Introduce nurse plants* **0 studies**

We captured no evidence for the effect of introducing nurse plants on naturally colonizing, focal peatland vegetation.

12.15 *Build artificial bird perches to encourage seed dispersal* **1 study**

Vegetation cover: One replicated, paired, controlled study in a peat swamp forest in Indonesia found that artificial bird perches had no significant effect on seedling abundance.

Introduce peatland vegetation

12.16.1 *Directly plant peatland mosses* **7 studies**

Survival: One study in Lithuania reported that 47 of 50 *Sphagnum*-dominated sods planted into a rewetted bog survived for one year.

Growth: Two before-and-after studies, in a fen in the Netherlands and bog pools in the UK, reported that mosses grew after planting.

Moss cover: Five before-and-after studies in a fen in the Netherlands and bogs in Germany, Ireland, Estonia and Australia reported that after planting mosses, the area covered by moss increased in at least some cases. The study in the Netherlands reported spread of planted moss beyond the introduction site. The study in Australia was controlled and reported that planted plots developed greater *Sphagnum* moss cover than unplanted plots.

12.16.2 *Directly plant peatland herbs* **5 studies**

Survival: Three replicated studies, in a fen meadow in the Netherlands and fens in the USA, reported that planted herbs survived over 2–3 years. However, for six of nine species only a minority of individuals survived.

Growth: Two replicated before-and-after studies, in a bog in Germany and fens in the USA, reported that individual planted herbs grew.

Vegetation cover: One replicated, controlled, before-and-after study in Canada found that planting herbs had no effect on moss, herb or shrub cover in created bog pools relative to natural colonization.

12.16.3 *Directly plant peatland trees/shrubs* **11 studies**

Survival: Eight studies (seven replicated) in peat swamp forests in Thailand, Malaysia and Indonesia and bogs in Canada reported that the majority of planted trees/shrubs survived over periods between 10 weeks and 13 years. One replicated study in a fen in the USA reported that most planted willow cuttings died within two years. One study in a peat swamp forest in Indonesia reported <5% survival of planted trees after five months, following unusually deep flooding.

Growth: Four studies (including two replicated, before-and-after) in peat swamp forests in Thailand, Indonesia and Malaysia reported that planted trees grew. One replicated before-and-after study in bogs in Canada reported that planted shrubs grew.

12.17.1 *Add mosses to peatland surface* **13 studies**

***Sphagnum* moss cover:** Eleven studies in bogs in the UK, Canada, Finland and Germany and fens in the USA reported that *Sphagnum* moss was present, after 1–4 growing seasons, in at least some plots sown with *Sphagnum*. Cover ranged from negligible to >90%. Six of these studies were controlled and found that there was more *Sphagnum* in sown than unsown plots. One additional study in Canada found that adding *Sphagnum* to bog pools did not affect *Sphagnum* cover.

Other moss cover: Four studies (including one replicated, randomized, paired, controlled, before-and-after) in bogs in Canada and fens in Sweden and the USA reported that mosses other than *Sphagnum* were present, after 2–3 growing seasons, in at least some plots sown with moss fragments. Cover ranged from 0 to 76%. In the fens in Sweden and the USA, moss cover was low (<1%) unless the plots were mulched, shaded or limed.

12.17.2 *Add mixed vegetation to peatland surface* **18 studies**

Characteristic plants: One replicated, randomized, paired, controlled, before-and-after study in a degraded bog (being restored as a fen) in Canada found that adding fen vegetation increased the number and cover of fen-characteristic plant species.

***Sphagnum* moss cover:** Seventeen replicated studies (five also randomized, paired, controlled, before-and-after) in bogs in Canada, the USA and Estonia reported that *Sphagnum* moss was present, after 1–6 growing seasons, in at least some plots sown with vegetation containing *Sphagnum*. Cover ranged from <1 to 73%. Six of the studies were controlled and found that *Sphagnum* cover was higher in sown than unsown plots. Five of the studies reported that *Sphagnum* cover was very low (<1%) unless plots were mulched after spreading fragments.

Other moss cover: Eight replicated studies (seven before-and-after, one controlled) in bogs in Canada, the USA and Estonia reported that mosses or bryophytes other than *Sphagnum* were present, after 1–6 growing seasons, in at least some plots sown with mixed peatland vegetation. Cover was <1–65%.

Vascular plant cover: Ten replicated studies in Canada, the USA and Estonia reported that vascular plants appeared following addition of mixed vegetation fragments to bogs. Two of the studies were controlled: one found that vascular plant cover was significantly higher in sown than unsown plots, but one found that sowing peatland vegetation did not affect herb cover.

12.18.1 *Introduce seeds of peatland herbs* — 10 studies

Germination: Two replicated studies (one also controlled, before-and-after) reported that some planted herb seeds germinated. In a bog in Germany three of four species germinated, but in a fen in the USA only one of seven species germinated.

Characteristic plants: Three studies (two controlled) in fen meadows in Germany and a peatland in China reported that wetland-characteristic or peatland-characteristic plants colonized plots where herb seeds were sown (sometimes along with other interventions).

Herb cover: Three before-and-after studies (one also replicated, randomized, paired, controlled) in a bog in New Zealand, fen meadows in Switzerland and a peatland in China reported that plots sown with herb seeds developed cover of the sown herbs (and, in New Zealand, greater cover than unsown plots). In China, the effect of sowing was not separated from the effects of other interventions. One replicated, randomized, paired, controlled study in a fen in the USA found that plots sown with herb (and shrub) seeds developed similar herb cover to plots that were not sown.

Overall vegetation cover: Of three replicated, controlled studies, one in a fen in the USA found that sowing herb (and shrub) seeds increased total vegetation cover. One study in a bog in New Zealand found that sowing herb seeds had no effect on total vegetation cover. One study in a fen meadow in Poland found that the effect of adding seed-rich hay depended on other treatments applied to plots.

Overall plant richness/diversity: Two replicated, controlled studies in fens in the USA and Poland found that sowing herb seeds had no effect on plant species richness (total or vascular). Two replicated, controlled, before-and-after studies in a bog in New Zealand and a fen meadow in Poland each reported inconsistent effects of herb sowing on total plant species richness.

12.18.2 *Introduce seeds of peatland trees/shrubs* — 5 studies

Germination: Two replicated studies in a bog in Germany and a fen in the USA reported germination of heather and willow seeds, respectively, in at least some sown plots.

Survival: One replicated study in a bog in Germany reported survival of some heather seedlings over two years. One replicated study in a fen in the USA reported that all germinated willow seedlings died within one month.

Shrub cover: Two studies (one replicated, randomized, paired, controlled) in bogs in New Zealand and Estonia reported that plots sown with shrub seeds, sometimes along with other interventions, developed greater cover of some shrubs than plots that were not sown: sown manuka or naturally colonizing heather (but not sown cranberry). One replicated, randomized, paired, controlled study in a fen in the USA found that plots sown with shrub (and herb) seeds developed similar overall shrub cover to unsown plots within two years.

Overall vegetation cover: Two replicated, randomized, paired, controlled studies in a bog in New Zealand and a fen in the USA reported that plots sown with shrub (and herb) seeds developed greater total vegetation cover than unsown plots after two years. One site comparison study in bogs in Estonia reported that sowing shrub seeds, along with fertilization, had no effect on total vegetation cover after 25 years.

Overall plant richness/diversity: One site comparison study in bogs in Estonia reported that sowing shrub seeds, along with fertilization, increased plant species richness. However, one replicated, randomized, paired, controlled study in a bog in New Zealand reported that plots sown with shrub seeds typically contained fewer plant species than plots that were not sown. One replicated, randomized, paired, controlled study in a fen in the USA found that sowing shrub (and herb) seeds had no effect on plant species richness.

Interventions: General habitat creation and restoration

12.1 Restore/create peatland vegetation (multiple interventions) Ⓑ Ⓕ Ⓢ

- **Nine studies** examined the effect of multiple restoration interventions (other than the moss layer transfer technique as defined in Section 12.2) on peatland vegetation. Six studies were in bogs[1,2,3,4,6,8] (one[8] being restored as a fen). One study was in a fen[7]. Two studies were in unspecified or mixed peatlands[5,9].
- **Plant community composition (3 studies):** One replicated, controlled, before-and-after study in the UK[1] reported that the overall plant community composition differed between restored and unrestored bogs. One replicated, controlled, site comparison study in Estonia[6] found that restored and natural bogs contained more similar plant communities than unrestored and natural bogs. However, one site comparison study in Canada[8] reported that after five years, bogs being restored as fens contained a different plant community to natural fens.
- **Characteristic plants (1 study):** One controlled study in a fen in France[7] reported that restoration interventions increased cover of fen-characteristic plants.
- **Moss cover (7 studies):** Five studies (including one replicated, paired, controlled, before-and-after) in bogs or other peatlands in the UK[1,4,5], Estonia[6] and Canada[9] found that restoration interventions increased total moss (or bryophyte[5,6]) cover. Two studies (one replicated and controlled) in bogs in the Czech Republic[2] and Estonia[6] reported that restoration interventions increased *Sphagnum* moss cover, but one replicated before-and-after study in bogs in the UK[4] reported no change in *Sphagnum* cover following intervention. Two site comparison studies in Canada[8,9] reported that after 1–15 years, restored areas had lower moss cover than natural fens.
- **Herb cover (5 studies):** Five studies (one replicated, paired, controlled, before-and-after) in bogs or other peatlands in the Czech Republic[2], the UK[4,5], Estonia[6] and Canada[9] reported that restoration interventions increased cover of herbaceous plants, including cottongrass[2,4,6] and other grass-like plants[5,9].
- **Overall vegetation cover (3 studies):** Three studies (one replicated, controlled, before-and-after) in bogs in the UK[1,4] and France[3] reported that restoration interventions increased overall vegetation cover.

Background

Peatland creation and restoration schemes can involve many different specific interventions. This section considers peatland creation and restoration using more than three separate interventions at once, such that it is difficult to attribute outcomes to any single intervention. Where three or fewer interventions have been used together in a study, results are reported elsewhere in the synopsis: under each intervention (but noting the influence of the others, where appropriate) or sometimes as a combined intervention (e.g. tree removal and rewetting in Section 3.4).

Tree-colonized describes peatlands that would not naturally contain trees. *Forested* or *swamps* are used to describe peatlands with natural tree cover. *Restored* refers to areas where restoration interventions have been applied (i.e. undergoing the process of restoration) rather than the state of those areas (i.e. whether they have been successfully restored).

Related interventions: restoration using the moss layer transfer technique, a specific combination of multiple interventions (Section 12.2).

A replicated, controlled, before-and-after study in 2007–2010 in a degraded blanket bog in England, UK (1) reported that areas restored using multiple interventions developed a different plant community to unrestored areas, and found that they had greater vegetation cover. All areas were initially bare peat. Three years after intervention, the overall plant community composition differed between restored and unrestored areas (data reported as a graphical analysis; difference not tested for statistical significance). Restored areas had developed greater cover than an unrestored area of total vegetation (60–88% vs 15%), mosses (13–25% vs 1%) and heather *Calluna vulgaris* (2–25% vs 1%). Heather cover was particularly high in plots covered with heather brash. Note that most of the vegetation cover in restored areas was the nurse grass (33–47% cover). In winter 2007/2008, four areas (bare gully sides) were restored by sowing grass seed as a nurse crop (41 kg/ha, mix of six species), fertilization (nitrogen-phosphorous-potassium, 250 kg/ha), liming (1 t/ha) and gully blocking (with stone or heather bales). Two areas were also covered with heather brash (including heather seeds) and one covered in geojute matting. One additional area was not restored (received no intervention). Before monitoring began, sheep were excluded from the entire bog. Vegetation cover was estimated before (summer 2007) and after (summer 2010) restoration, in thirty 2 x 2 m quadrats/area.

A study in 1999–2007 in a historically mined bog in the Czech Republic (2) reported that following multiple restoration interventions, bare peat was colonized by vegetation including *Sphagnum* moss, cottongrasses *Eriophorum* spp. and beaked sedge *Carex rostrata*. These results were not tested for statistical significance. Of the bare peat present one year after restoration began, 30% was covered by vegetation seven years later. Over this time, *Sphagnum* cover increased from <2% to 8%. Cover also increased of common cottongrass *Eriophorum angustifolium*, sheathed cottongrass *Eriophorum vaginatum* and beaked sedge (data reported as maps). Between 1999 and 2004, a historically mined bog (with some remnant vegetation in drainage ditches) was subjected to multiple restoration interventions: rewetting by blocking drainage ditches, excavating shallow 10 x 10 m basins, planting cottongrasses and beaked sedge, sowing *Sphagnum* moss, mulching (both sown and unsown areas) with sedge cuttings, removing trees and stabilizing the peat surface with tree trunks. In 2000 and 2007, maps were made of where each plant species was dominant (or co-dominant).

A study in 2003–2008 in a historically mined raised bog in France (3) reported that following restoration by multiple interventions, vegetation cover increased. No statistical tests were carried out. When intervention began, total vegetation cover was 14%. Five years later, it had increased to 51%. Restoration of 0.2 ha of drained, nutrient-enriched bog began in 2003. The top 20 cm of peat were removed, leaving an uneven surface. The stripped peat was used to build embankments and block a drainage ditch, rewetting the area. It was then planted (with sheathed cottongrass *Eriophorum vaginatum*, common cottongrass *Eriophorum angustifolium* and *Sphagnum* mosses) and mulched with straw. Annually between 2003 and 2008, vegetation cover was estimated in six 1 m^2 quadrats across the restored area.

A replicated before-and-after study in 2002–2010 in five degraded blanket bogs in England, UK (4) reported that following multiple restoration interventions, vegetation cover increased. These results are not based on tests of statistical significance. There were increases in total vegetation cover (before intervention: 2–14%; after eight years: 162–200%), cover of cottongrass *Eriophorum* spp. (before: 0%; after: 21–49%), cover of planted shrubs (before: 0%; after: 1–24%) and cover of mosses other than *Sphagnum* (before: 0%; after: 20–50%). The increase in *Sphagnum* moss cover was much smaller (before: 0%; after: <1%). Note that cover of nurse grasses, after eight years, was 27–50%. Between 2002 and 2008, five areas of bare peat were treated with multiple restoration interventions: building fences to exclude livestock, ditch blocking to raise the water table, liming, fertilization, adding brash (heather *Calluna vulgaris*) or a fibre mesh, sowing grass seeds, planting grass and shrub plants and spreading *Sphagnum* fragments. Vegetation cover was estimated before restoration began (2002), then annually between 2003 and 2010, in 4–44 variously sized quadrats/bog.

A replicated before-and-after study in 2008–2011 in a tree-colonized peatland in Scotland, UK (5) reported that plots restored using multiple interventions developed herb and bryophyte cover. After 2–3 years, restored plots had 42% cover of rush *Juncus* spp., 21% cover of bryophytes, 10% cover of devil's bit scabious *Succisa pratensis* and 20% cover of other vegetation (including heather *Calluna vulgaris*, grasses, sedges and other herbs). Cover of devil's bit scabious did not significantly differ between grazed and ungrazed plots (data not reported). In October 2008, eight 16 m^2 plots were restored by cutting and removing all conifer trees and sowing seeds of devil's bit scabious. In four plots, conifer brash was burned after tree removal. Four plots were fenced to exclude deer. In August 2011, vegetation cover was visually estimated in five random 2 x 2 m quadrats/plot.

A replicated, controlled, site comparison study in 2012–2014 in a historically mined bog in Estonia (6) found that restoration by multiple interventions increased cover of bryophytes and vascular plants, and created a plant community more like the natural donor bog. After 1–2 years, restored plots had greater cover than an unrestored plot of all bryophytes combined (52–65% vs <1%), *Sphagnum* mosses (50–54% vs <1%) and vascular plants (17–23% vs 11%). Sheathed cottongrass *Eriophorum vaginatum* and sedge *Carex* sp. were present in at least one restored plot (cover <1%), but not in the unrestored plot. After two years, the overall plant community in restored plots was 40–67% similar to the unmined donor bog, compared to 21–29% similarity between the unrestored plot and donor bog. In spring 2012, three plots of almost-bare peat were restored by reprofiling (top 20 cm of peat pushed into ridges around the plot), rewetting (blocking a drainage ditch), adding plant fragments (mostly *Sphagnum* mosses) from the surface of a nearby bog and mulching with straw. One adjacent plot received no intervention. In June and

September 2013 and 2014, vegetation cover was estimated in ten 50 x 50 cm quadrats in each plot and the donor bog.

A controlled study in 1997–2014 in a degraded fen in France (7) reported that following restoration by multiple interventions, plant species richness and cover of fen-characteristic plants increased. No statistical tests were carried out. Fifteen years after intervention, there were 20 plant species in the fen (vs 12 when intervention began). Cover of fen-characteristic plants, including sedges *Carex* spp., was 50% (vs 2% when intervention began and vs 12% in an unrestored area of the fen). Restoration of a drained, abandoned, overgrown fen began in 1997–1999. The fen was rewetted by remeandering an adjacent river. Willow shrubs were cut and removed. Existing herbaceous vegetation and surface peat were shredded. Each summer between 1999 and 2004, the fen was grazed by horses. From 2004, a four year cycle of grazing-rest-mowing-rest was implemented in a mosaic across the fen. Between 1999 and 2014, cover of plant community types was estimated in the managed area (in four quadrats along a transect) and adjacent unmanaged fen (details not reported).

A site comparison study in 2008–2014 in a historically mined bog in Quebec, Canada (8) reported that an area restored using multiple interventions developed a different plant community to, with less vegetation cover than, nearby natural fens. These results were not tested for statistical significance. Note that the aim of this study was to create a fen, as the post-mining peat chemistry was more like a fen than a bog. Five years after intervention, the overall plant community composition of the restored area was different from three nearby natural fens (data reported as a graphical analysis). In the restored area, *Sphagnum* moss cover was only 3% (vs 15–25% in natural fens) and other moss cover only 3% (vs 12–55%). Vascular plant cover was only 26% (vs 59–86%), although the dominant species were similar in the restored fen and its donor fen. In winter 2009/2010, part of a historically mined bog (abandoned for nine years) was restored by rewetting (blocking drainage ditches with peat), excavating peat basins (removing surface peat and building embankments), sowing vegetation fragments from a moss-dominated donor fen, and mulching with straw. Vegetation cover was estimated in 2008 (donor fen; in 16 quadrats along a transect) or 2014 (restored area; in five 25 m^2 plots).

A replicated, paired, controlled, before-and-after, site comparison study in five historically mined peatlands in Canada (9) found that restoration by multiple interventions increased cover of mosses, grass-like plants and vascular plants, but not shrubs. Restored and unrestored plots were initially bare peat. After 1–15 years, restored sites had significantly greater cover than unrestored sites of mosses (38 vs 3%), grass-like plants (22 vs 5%) and total vascular plants (33 vs 11%), but there was no significant difference in shrub cover (9 vs 3%). Relative to natural, undisturbed sites, restored sites had lower cover of mosses (38 vs 77%), shrubs (9 vs 27%) and total vascular plants (33 vs 44%), but higher cover of grass-like plants (22 vs 3%). Five degraded peatlands were restored (dates unclear) using a mixture of techniques. All received fresh vegetation fragments from the surface of natural peatlands and were mulched with straw. Some sites were levelled, rewetted and/or fertilized. Summer vegetation cover was estimated in restored sites after 1–15 years. For each restored site, vegetation cover of a natural peatland was estimated either before restoration or in separate untreated areas after restoration.

(1) Anderson P., Worrall P., Ross S., Hammond G. & Keen A. (2011) *United Utilities Sustainable Catchment Management Programme Volume 3: The Restoration of Highly Degraded Blanket Bog*. Penny Anderson Associates Project Report.

(2) Horn P. (2012) *Ekologie rašelinišť na Šumavě* (*Ecology of peat bogs in Šumava*; in Czech). PhD Thesis. University of South Bohemia.

(3) Pôle-Relais Tourbières (2013) *Restauration des milieux tourbeux: essais de techniques de restauration des casiers d'exploitation de la tourbières "sur les Seignes" à Frambouhans* (Restoration of peatlands: trials of techniques for the restoration of the bogs on the Seignes in Frambouhans; in French). Retours d'expériences, Pôle-Relais Tourbières Français.

(4) Proctor S., Buckler M., Walker J.S. & Maskill R. (2013) *Vegetation Recovery on Bare Peat after Restoration Intervention: An Analysis of Nine Years of Monitoring Data in the Dark Peak Moorlands (2003 - 2012)*. Moors for the Future Research Report.

(5) Kirkland P. (2014) *Experimental Marsh Fritillary Habitat Restoration Project Avongovie, Islay*. Scottish Natural Heritage Commissioned Report 544.

(6) Karofeld E., Müür M. & Vellak K. (2016) Factors affecting re-vegetation dynamics of experimentally restored extracted peatland in Estonia. *Environmental Science and Pollution Research*, 23, 13706–13717.

(7) Pôle-Relais Tourbières (2016) *La gestion par fauche des milieux humides de la réserve naturelle nationale du lac de Remoray* (Management by mowing in the wetlands of the Remoray Lake National Nature Reserve; in French). Retours d'expériences, Pôle-Relais Tourbières Français.

(8) Rochefort L., LeBlanc M.-C., Bérubé V., Hugron S., Boudreau S. & Pouliot R. (2016) Reintroduction of fen plant communities on a degraded minerotrophic peatland. *Canadian Journal of Botany*, 94, 1041–1051.

(9) Strack M., Cagampan J., Hassanpour Fard G., Keith A.M., Nugent K., Rankin T., Robinson C., Strachan I.B., Waddington J.M. & Xu B. (2016) Controls on plot-scale growing season CO_2 and CH_4 fluxes in restored peatlands: do they differ from unrestored and natural sites? *Mires and Peat*, 17, Article 5.

N.B. Results from (2) are also reported in: Horn P. & Bastl M. (2012) Restoration of the mined peatbog Soumarský Most. Pages 83–85 in: I. Jongepierová, P. Pešout, J.W. Jongepier & K. Prach (eds.) *Ecological Restoration in the Czech Republic*. Nature Conservation Agency of the Czech Republic, Prague.

12.2 Restore/create peatland vegetation using the moss layer transfer technique

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- **Four studies** examined the effect on peatland vegetation of restoration using the moss layer transfer technique (as defined in the background section). All four studies were based on bogs in Canada. Three studies[1,3,4] were based on one experimental set-up that was included in the other, larger study[2].
- **Plant community composition (2 studies):** One replicated study in bogs in Canada[2] reported that the majority of restored areas developed a community of bog-characteristic plant species within 11 years. One controlled, before-and-after study in a bog in Canada[3] reported that a restored area (included in the previous study) developed a more peatland-characteristic plant community over time, and relative to an unrestored area.
- **Vegetation cover (2 studies):** Two controlled studies in one bog in Canada[1,4] reported that a restored area had greater moss or bryophyte cover (including *Sphagnum*) than an unrestored area after 4–8 years. The restored area also had greater herb cover (including cottongrass), but less shrub cover, than the unrestored area. One of the studies[1] reported that vegetation in the restored area became more similar to local natural bogs.
- **Overall plant richness/diversity (1 study):** One controlled, before-and-after study in a bog in Canada[3] reported that a restored area contained more plant species than an unrestored area.

Background

The moss layer transfer technique (Quinty & Rochefort 2003; Rochefort *et al.* 2013) combines multiple interventions to restore peatlands: (1) Rewetting the peat, for example by blocking drainage ditches or building water-retaining ridges. (2) Reprofiling the peat i.e. clearing and flattening the peat surface. Surface peat may be pushed into water-retaining ridges around or across the peatland, achieving steps 1 and 2 in one go. (3) Spreading plant fragments collected from the surface (top 10 cm) of a nearby bog. (4) Adding a straw mulch to provide shade and keep the surface of the peat moist. (5) Fertilizing with phosphorous to stimulate growth of nurse plants such as haircap moss. To be included as evidence in this section, studies must have tested all five steps in combination – although fertilizing is an optional step that is not always appropriate (e.g. Karofeld *et al*. 2016).

The moss layer transfer technique is typically used to restore bogs damaged by peat extraction. Whether it is more appropriate to introduce vegetation from bogs or fens will depend on the chemistry of the peat remaining in the degraded peatland (Quinty & Rochefort 2003). In this section, *restored* refers to areas where restoration interventions have been applied (i.e. undergoing the process of restoration) rather than the state of those areas (i.e. whether they have been successfully restored).

CAUTION: Collecting plant fragments damages the donor site, although rapid recovery has been reported (Rochefort & Campeau 2002).

Related interventions: restoration using other combinations of multiple interventions (Section 12.1). Interventions within the moss layer transfer technique, tested individually: rewetting (Section 8.1); reprofiling (Section 12.5); sowing vegetation (Section 12.17); adding mulch after planting (Section 13.4); fertilizing to complement planting (Section 13.2).

Karofeld E., Müür M. & Vellak K. (2016) Factors affecting re-vegetation dynamics of experimentally restored extracted peatland in Estonia. *Environmental Science and Pollution Research*, 23, 13706–13717.

Quinty F. & Rochefort L. (2003) *Peatland Restoration Guide, Second Edition*. Canadian *Sphagnum* Peat Moss Association and New Brunswick Department of Natural Resources and Energy. Quebec, Canada.

Rochefort L. & Campeau S. (2002) Recovery of donor sites used for peatland restoration. Pages 244–251 in: G. Schmilewski & L. Rochefort (eds.) *IPS Symposium Proceedings: Peat in horticulture – quality and environmental challenges*. International Peat Society, Jyväskylä, Finland.

Rochefort L., Isselin-Nondedeu F., Boudreau S. & Poulin M. (2013) Comparing survey methods for monitoring vegetation change through time in a restored peatland. *Wetlands Ecology and Management*, 21, 71–85.

A controlled, before-and-after, site comparison study in 1999–2003 in a historically mined bog and 92 natural bogs in Quebec, Canada (1) reported that an area restored using the moss layer transfer technique developed greater vegetation cover than an unrestored area, and that vegetation in the restored area was more similar to that of natural bogs. These results were not tested for statistical significance. After four years, the restored area had 49% moss cover (vs unrestored: 2%; natural: 85%), 19% herb cover (unrestored: 8%; natural: 19%) and 8% shrub cover (unrestored: 13%; natural: 51%). In autumn 1999, 8.4 ha of historically mined bog were restored by levelling the surface, rewetting (blocking drainage ditches and building embankments), adding plant fragments from the surface of a nearby natural peatland, straw mulching, and phosphorous fertilization. An adjacent 3.1 ha was not restored. In August 2003, vegetation cover was recorded in 3 x 8 m quadrats: 32 in the restored area and 15 in the unrestored area. Vegetation cover in 92 nearby natural (unmined) bogs was recorded in 2000. This study was based on the same experimental set-up as (3) and (4).

A replicated study in 1997–2012 in 12 historically mined bogs in Canada (2) reported that most areas restored using the moss layer transfer technique developed a community of bog-characteristic plant species within 4–11 years. These results are not based on tests of statistical significance. Of 34 restored areas, 23 had developed a community of bog-characteristic plants (data reported as a graphical analysis). These areas were dominated by red bog moss *Sphagnum rubellum* (37% cover) and cottongrasses *Eriophorum* spp. (4–20% cover). Eleven areas did not develop this characteristic community. Eight were dominated by haircap moss *Polytrichum strictum* (60% cover). Three areas developed high cover of bare peat (52% cover), birch *Betula* sp. (12% cover) and lichens (4% cover). Between 1997 and 2004, 34 areas in 12 historically mined bogs were restored by levelling the peat surface, rewetting (blocking drainage ditches), adding *Sphagnum*-dominated vegetation fragments and mulching with straw. Some areas were also fertilized with phosphorous. Vegetation cover was estimated 4–11 years after intervention: vascular plants in 1 x 1 m quadrats (4–128/area) and bryophytes in 25 x 25 cm quadrats (20–640/area). This study included the site restored in (1), (3) and (4).

A controlled, before-and-after study in 1998–2007 in a historically mined bog in Quebec, Canada (3) reported that an area restored using the moss layer transfer technique developed a more peatland-characteristic plant community than an unrestored area, with higher richness and diversity of characteristic plants (and higher overall plant species richness). These results were not tested for statistical significance. Before intervention, both areas contained a similar community of weedy, shrubby and forest plants. Over eight years, the restored area developed a community of peatland-characteristic plants but the unrestored area did not. Red bog moss *Sphagnum rubellum* became particularly abundant in the restored area (data reported as graphical analyses). After eight years, the restored area contained more plant species than the unrestored area (21 vs 17), more peatland-characteristic plant species (11 vs 3; before intervention: 1) and more wetland-characteristic plant species (2 vs 0; before intervention: 0). The restored area also had higher diversity of the characteristic species than the unrestored area, but lower total plant diversity (data reported as diversity indices). In 1999, 8.4 ha of historically mined bog were restored by levelling, rewetting (building embankments and blocking drainage ditches), adding *Sphagnum*-dominated vegetation fragments and mulching with straw. Fertilizer was added the following summer. In the same peatland, 3.1 ha were not restored. In 1998 and 2001–2007, cover of every plant species was measured using rods dropped at over 7,000 points along transects. This study was based on the same experimental set-up as (1) and (4).

A controlled, before-and-after study in 1999–2007 in a historically mined bog in Quebec, Canada (4) reported that an area restored using the moss layer transfer technique had greater cover of bryophytes and herbs, and lower tree/shrub cover, than an unrestored area. These results were not tested for statistical significance. After eight years, the restored area had total bryophyte cover of 79% (vs 19% in the unrestored area), *Sphagnum* moss cover of 60% (unrestored: 0%), total herb cover of 76% (unrestored: 18%) and sheathed cottongrass *Eriophorum vaginatum* cover of 50% (unrestored: 5%). In contrast, the restored area had only 5% tree/shrub cover, compared to 23% in the unrestored area. Before restoration, vegetation cover was low (e.g. bryophytes <15%, herbs <20%) and similar across areas later restored and unrestored. In 1999, 8.4 ha of historically mined bog were restored by levelling, rewetting (building embankments and blocking drainage ditches), adding *Sphagnum*-

dominated vegetation fragments and mulching with straw. Fertilizer was added the following summer. In the same peatland, 3.1 ha were not restored. In July 1999 (before restoration) and biannually between 2001 and 2007, plant species were recorded at approximately 5,700 points across the bog. Similar results were obtained when cover was visually estimated in forty-three 3 x 8 m quadrats. This study was based on the same experimental set-up as (1) and (3).

(1) Mazerolle M.J., Poulin M., Lavoie C., Rochefort L., Desrochers A. & Drolet B. (2006) Animal and vegetation patterns in natural and man-made bog pools: implications for restoration. *Freshwater Biology*, 51, 333–350.
(2) González E., Rochefort L., Boudreau S., Hugron S. & Poulin M. (2013) Can indicator species predict restoration outcomes early in the monitoring process? A case study with peatlands. *Ecological Indicators*, 32, 232–238.
(3) Poulin M., Andersen R. & Rochefort L. (2013) A new approach for tracking vegetation change after restoration: a case study with peatlands. *Restoration Ecology*, 21, 363–371.
(4) Rochefort L., Isselin-Nondedeu F., Boudreau S. & Poulin M. (2013) Comparing survey methods for monitoring vegetation change through time in a restored peatland. *Wetlands Ecology and Management*, 21, 71–85.

Interventions: Modify physical habitat only

12.3 Fill/block ditches to create conditions suitable for peatland plants Ⓑ Ⓕ Ⓢ

- **Three studies** examined the effect of filling or blocking ditches (without planting) on peatland vegetation within them. Two studies were in bogs[1,2] and one was in a fen[3].
- **Vegetation cover (3 studies):** Two studies in a bog in the UK[2] and a fen in the USA[3] reported that blocked or filled ditches were colonized by herbs and bryophytes within 2–3 years. In the USA[3], vegetation cover (total, bryophyte, forb, grass and sedge) was restored to natural, undisturbed levels. One replicated study in bogs in the UK[1] reported that plants had not colonized blocked gullies after six months.
- **Overall plant richness/diversity (1 study):** One site comparison study in a fen in the USA[3] found that a filled ditch contained more plant species than adjacent undisturbed fen, after two years.

Background

Many threats can contribute to the formation of ditches or channels in peatlands (Evans *et al.* 2005). Deep channels (gullies) may be eroded by humans or livestock repeatedly using set trails and/or by heavy rainfall events. Erosion can be increased by burning and acid rain. Ditches may be deliberately dug to drain peatlands for agriculture, forestry or mining. Peatland vegetation cannot establish in ditches that contain deep water or are regularly disturbed by flowing water.

This section considers growth of peatland vegetation *within* filled or blocked ditches. Ditches could be completely filled to immediately create a surface for plants to grow on. Alternatively, ditches could be blocked with dams: creating shallow pools, encouraging peat deposition and eventually creating new surfaces on which peatland plants can grow. This would mimic natural revegetation processes (Evans *et al.* 2005).

Related intervention: rewetting larger areas of peatland, including by blocking ditches (Section 8.1).

Evans M., Allott T., Crowe S. & Liddaman L. (2005) Feasible locations for gully blocking. Pages 27–76 in: M. Evans, T. Allott, J. Holden, C. Flitcroft & A. Bonn (eds.) *Understanding Gully Blocking in Deep Peat*. Moors for the Future Research Report 4.

A replicated study in 2003–2004 in two degraded blanket bogs in England, UK (1) reported that gullies blocked with dams had no vegetation cover after approximately six months. In late 2003, 389 blockages were installed along 16 gullies. A mixture of blocking techniques was used: wooden fences, plastic fences, stone walls or stacked hessian sacks. Vegetation cover was visually estimated between May and July 2004.

A replicated study in 2006 in a blanket bog in England, UK (2) reported that drainage ditches blocked with peat developed some vegetation cover but were mostly bare peat. All five blocked drainage ditches developed some vegetation cover, although total vegetation cover was <50% in four of them. Across all five ditches, cover of common cottongrass *Eriophorum angustifolium* was 5–30%.Cover of *Sphagnum* moss was <1–20%. One ditch (also recently burned) had 60% cover of heather *Calluna vulgaris*. The study noted correlations between vegetation cover, slope and soil/water chemistry. In January 2003, five ditches were blocked with peat sods. Vegetation cover was estimated in spring 2006 (A. Armstrong pers. comm.).

A site comparison study in 2009–2011 in a fen in Michigan, USA (3) found that a ditch filled with peat spoil developed similar vegetation cover to undisturbed areas of the fen, but contained more plant species. After two years, the filled ditch and adjacent areas of undisturbed fen had similar cover of total vegetation (165 vs 180%), sedges (81 vs 80%), grasses (15 vs 10%), forbs (33 vs 20%) and bryophytes (29 vs 40%). However, there were more plant species in the filled ditch (49 species) than undisturbed fen (40 species). Results after one year were similar, except that the ditch had lower total vegetation cover than undisturbed fen (126 vs 188%) and lower bryophyte cover (18 vs 40%). In 2009, a ditch (dug in 2007 as a fire break) was filled with adjacent spoil (still moist and containing fen plant seeds). In 2010 and 2011, vegetation cover was recorded in sixty 1 m^2 quadrats along the length of the ditch: 20 within it and 20 on either side. These were placed in areas not sown with additional seeds.

(1) Evans M., Allott T., Crowe S. & Liddaman L. (2005) Feasible locations for gully blocking. Pages 27–76 in: M. Evans, T. Allott, J. Holden, C. Flitcroft & A. Bonn (eds.) *Understanding Gully Blocking in Deep Peat*. Moors for the Future Research Report 4.
(2) Armstrong A., Holden J. & Stevens C. (2008) *The Differential Response of Vegetation to Grip Blocking*. Report to North Pennines AONB.
(3) Bess J.A., Chimner R.A. & Kangas L.C. (2014) Ditch restoration in a large Northern Michigan fen: vegetation response and basic porewater chemistry. *Ecological Restoration*, 32, 260–274.

12.4 Excavate pools

B F S

- **Two studies** examined the effect of excavating pools (without planting) on peatland vegetation. Both studies were based on the same experimental set-up in bogs in Canada.
- **Plant community composition (1 study):** One replicated, before-and-after, site comparison study in bogs in Canada[2] reported that excavated pools were colonized by some peatland

vegetation over 4–6 years, but contained different plant communities to natural pools. In particular, cattail was more common in created pools.

- **Vegetation cover (1 study):** One replicated, before-and-after, site comparison study in bogs in Canada[1] reported that after four years, created pools had less cover than natural pools of *Sphagnum* moss, herbs and shrubs.
- **Overall plant richness/diversity (1 study):** One replicated, before-and-after, site comparison study in bogs in Canada[2] reported that after six years, created pools contained a similar number of plant species to natural pools.

Background

Peatlands may contain permanent pools which contribute to habitat diversity. However, these pools can be lost following drainage, peat harvesting or drought (Beadle *et al.* 2015). Pools may also be filled in naturally as vegetation develops and peat accumulates, so they may need to be re-excavated to maintain certain communities (van Diggelen *et al.* 1996). This section considers excavating pools in peatlands, then leaving vegetation to naturally recolonize.

Related interventions: rewetting, which may need to be done to fill created pools (Section 8.1); introduce peatland vegetation, including introductions into created pools (Sections 12.16 and 12.17).

Beadle J.M., Brown L.E. & Holden J. (2015) Biodiversity and ecosystem functioning in natural bog pools and those created by rewetting schemes. *WIREs Water*, 2, 65–84.

van Diggelen R., Molenaar W.J. & Kooijman A.M. (1996) Vegetation succession in a floating mire in relation to management and hydrology. *Journal of Vegetation Science*, 7, 809–820.

A replicated, before-and-after, site comparison study in 1999–2003 in four bogs in eastern Canada (1) found that created pools developed vegetation cover within four years, but reported that this remained lower than cover in and around natural pools. Initially, the created pools contained no vegetation. After four years, *Sphagnum* moss cover had increased to 9% (vs 80% in natural pools), herb cover had increased to 5% (natural: 10%) and shrub cover had increased to 5% (natural: 27%). Comparisons with natural pools were not tested for statistical significance. In 1999, four 6 x 8.5 m pools were created in one historically mined bog by excavating and rewetting (blocking drainage ditches and building embankments). No vegetation was introduced to these, although the surrounding site was sown with bog vegetation fragments. In 2003, vegetation cover was recorded in 36 quadrats/pool, each 30 x 30 cm, along six bank-to-bank transects. Vegetation cover of 70 natural pools, in unmined parts of nearby bogs, was recorded in 1999 and 2000. This study was based on the same experimental set-up as (2).

A replicated, before-and-after, site comparison study in 1999–2005 in seven bogs in Quebec, Canada (2) reported that created pools developed a different plant community to natural pools, but with similar species richness. After six years, the overall composition of the plant community differed between created and natural pools (data reported as a turnover index and graphical analysis). In particular, cattail *Typha latifolia* was more frequent in created pools (occurring in 69% of quadrats) than natural pools (0% of quadrats). *Sphagnum* mosses, *Eriophorum* cottongrasses and *Carex* sedges were sometimes more abundant in restored pools and sometimes less abundant, depending on the species (see original paper). Created and natural pools both contained 24 plant species/0.5 m^2. In 1999, four 6 x 12 m pools were created in a historically mined bog by excavating and rewetting (blocking drainage ditches and building embankments). In 2005, cover of every plant species was estimated in 0.5 m^2 quadrats situated on pool margins: 12 quadrats around the created pools and 30 around pools in each of six natural, unmined bogs. This study was based on the same experimental set-up as (1).

(1) Mazerolle M.J., Poulin M., Lavoie C., Rochefort L., Desrochers A. & Drolet B. (2006) Animal and vegetation patterns in natural and man-made bog pools: implications for restoration. *Freshwater Biology*, 51, 333–350.
(2) Fontaine N., Poulin M. & Rochefort L. (2007) Plant diversity associated with pools in natural and restored peatlands. *Mires and Peat*, 2, Article 6.

12.5 Reprofile/relandscape peatland

Ⓑ Ⓕ S

- **One study** examined the effect of reprofiling/relandscaping peatlands (without planting) on peatland vegetation. The study was in degraded bogs (being restored as fens).
- **Plant community composition (1 study):** One site comparison study in Canada[1] reported that after five years, reprofiled (and rewetted) bogs contained a different plant community to nearby natural fens.
- **Vegetation cover (1 study):** The same study[1] reported that after five years, reprofiled (and rewetted) bogs had lower vegetation cover (*Sphagnum* moss, other moss and vascular plants) than nearby natural fens.

Background

Large scale relandscaping of a degraded peatland may create a more suitable environment for vegetation growth. In particular, local moisture levels could be raised by reprofiling the peat surface into basins (removing <30 cm of surface peat, and pushing this into ridges) or flat terraces (removing mounds that are too dry for colonization). Steep gully sides or eroding slopes can be reprofiled into shallower, more stable slopes.

CAUTION: Heavy machinery used for landscaping may churn and compress the peat soil, damaging its structure. Removing surface peat from bogs may expose fen peat, which has different chemical properties to bog peat and will not (in the short term) support bog vegetation (Lindsay & Clough 2016).

Related interventions: rewetting, without altering the level of the peat surface (Section 8.1); removing upper layer of peat/soil with no other landscaping (Section 12.7); reprofiling/relandscaping before planting (Section 13.9).

Lindsay R.A. & Clough J. (2016) *A Review of the Influence of Ombrotrophic Peat Depth on the Successful Restoration of Bog Habitat*. Scottish Natural Heritage Commissioned Report 925.

A site comparison study in 2008–2014 in a historically mined bog in Quebec, Canada (1) reported that areas restored by creating terraces and embankments (and raising the water table) developed a different plant community to nearby natural fens, with less vegetation cover. These results were not tested for statistical significance. Note that the aim of this study was to create a fen, as the post-mining peat chemistry was more like a fen than a bog. Five years after intervention, the overall plant community composition of the restored areas was different from three nearby natural fens (data reported as a graphical analysis). In the restored areas, *Sphagnum* moss was absent (vs 15–25% in natural fens), other moss cover only 1% (vs 12–55%) and vascular plant cover only 24% (vs 59–86%). In winter 2009/2010, parts of a historically mined bog (abandoned for nine years) were reprofiled (by pushing the top 30 cm of degraded peat into embankments) and rewetted (by blocking drainage ditches). The study does not distinguish between the effects of these interventions. Vegetation cover was estimated in 2008 (donor fen; in 16 quadrats along a transect) or 2014 (restored areas; in five 25 m^2 plots).

(1) Rochefort L., LeBlanc M.-C., Bérubé V., Hugron S., Boudreau S. & Pouliot R. (2016) Reintroduction of fen plant communities on a degraded minerotrophic peatland. *Canadian Journal of Botany*, 94, 1041–1051.

12.6 Roughen peat surface to create microclimates

Ⓑ Ⓕ S

- We captured no evidence for the effect of roughening the peat surface to create microclimates (without planting afterwards) on peatland vegetation.

Background

Exposed bare peat can be too dry for peatland vegetation to naturally recolonize. Roughening the peat surface (e.g. by ploughing or driving over it, or by adding peat blocks) could benefit recolonization in two ways. First, it could create more suitable (sheltered and moist) microclimates for plant colonisation within low areas or depressions. Although raised areas might be less suitable for peatland vegetation initially, the idea is that peatland vegetation can spread from the depressions once they have been colonized. Second, disturbing the peatland surface will also break up any hard crust or loose peat that has developed, both of which can limit water supply to the vegetation above.

Related interventions: rewetting large areas of peatland by raising the water table (Section 8.1); roughen peat surface, by creating mounds or hollows, before planting (Section 13.10).

12.7 Remove upper layer of peat/soil

B Ⓕ S

- **Ten studies** examined the effect of removing the upper layer of peat or soil (without planting afterwards) on peatland vegetation. Nine studies were in fens or fen meadows[1,2,4,5,6,7,8,9,10] and one was in an unspecified peatland[3].
- **Plant community composition (6 studies):** Five studies (including one replicated, randomized, paired, controlled) in a peatland in the USA[3] and fens or fen meadows in the Netherlands[5,10] and Poland[7,8] reported that plots stripped of topsoil developed plant communities with a different composition to those in unstripped peatlands. In one study[10], the effect of stripping was not separated from the effect of rewetting. Two studies in fen meadows in Germany[2] and Poland[7] reported that the depth of soil stripping affected plant community development.
- **Characteristic plants (5 studies):** Four studies in fen meadows in Germany[2,4] and the Netherlands[10], and a peatland in the USA[3], reported that stripping soil increased cover of wetland-characteristic[2,3] or peatland-characteristic[4,10] plants after 4–13 years. In the Netherlands[10], the effect of stripping was not separated from the effect of rewetting. One

replicated site comparison study in fens in Belgium and the Netherlands[9] found that stripping soil increased fen-characteristic plant richness.

- **Herb cover (4 studies):** Three studies (including one replicated, paired, controlled) in fens or fen meadows in Germany[2], the UK[6] and Poland[8] found that stripping soil increased cover of rushes[2,6,8], reeds[8] or sedges[2,6] after 2–6 years. However, one controlled study in a fen meadow in the Netherlands[1] reported that stripping soil had no effect on sedge or bentgrass cover after five years. Two controlled studies in a fen meadow in the Netherlands[1] and a fen in the UK[6] found that stripping soil reduced purple moor grass cover for 2–5 years.
- **Vegetation structure (3 studies):** Two studies in fens or fen meadows in the Netherlands[1,9] and Belgium[9] found that stripping soil reduced vegetation biomass (total[1] or herbs[9]) for up to 18 years. One replicated, randomized, paired, controlled study in a peatland in the USA[3] found that stripping soil had no effect on vegetation biomass after four years.
- **Overall plant richness/diversity (6 studies):** Three studies (including one replicated, paired, controlled) in fens or fen meadows in the UK[6], Belgium[9] and the Netherlands[9,10] reported that stripping soil increased total plant species richness over 2–18 years. In one study[10], the effect of stripping was not separated from the effect of rewetting. One replicated, controlled study in a fen in Poland[8] found that stripping soil had no effect on plant species richness after three years. One replicated, randomized, paired, controlled study in a peatland in the USA[3] found that stripping soil increased plant species richness and diversity, after four years, in one field but decreased it in another. One replicated study in a fen meadow in Poland[7] reported that plant species richness increased over time, after stripping soil.

Background

Damaged peatlands may be covered by a layer of non-peat soil. Alternatively, the surface peat may contain excess nutrients or a seed bank of undesirable plants (e.g. left over from agricultural use), may be too acidic (e.g. as a result of atmospheric deposition), or may be covered in a hard crust or very loose peat (making plant establishment and growth more difficult). Fens will naturally develop into bogs as peat accumulates, but this change is not always desirable.

The upper layer of peat or soil (and any vegetation on it) could be removed from damaged peatlands, creating a new bare peat surface for colonization. This surface may have fewer nutrients, no undesirable seed bank, and often wetter and less acidic peat since the surface is closer to the water table (Grootjans *et al.* 2002). Stripping surface peat can reverse the development of fens into bogs.

CAUTION: Soil stripping may be unsuitable for wetter peatlands as heavy machinery involved may churn and compress the peat soil. Stripping surface peat from bogs may expose fen peat, which has different chemical properties to bog peat and will not (in the short term) support bog vegetation (Lindsay & Clough 2016).

Related interventions: rewetting (Section 8.1); reprofile/relandscape peatland e.g. by building ridges or embankments (Section 12.5); bury upper layer of peat or soil (Section 12.8); disturb peatland surface, but without removing peat/soil (Section 12.9); remove soil before planting (Section 13.11); interventions to control vegetation without also removing peat (Chapter 9).

Grootjans A.P., Bakker J.P., Jansen A.J.M. & Kemmers R.H. (2002) Restoration of brook valley meadows in the Netherlands. *Hydrobiologia*, 478, 149–170.

Lamers L.P.M., Vile M.A., Grootjans A.P., Acreman M.C., van Diggelen R., Evans M.G., Richardson C.J., Rochefort L., Kooijman A.M., Roelofs J.G.M. & Smolders A.J.P. (2015) Ecological restoration of rich fens in

Europe and North America: from trial and error to an evidence-based approach. *Biological Reviews*, 90, 182–203.

Lindsay R.A. & Clough J. (2016) *A Review of the Influence of Ombrotrophic Peat Depth on the Successful Restoration of Bog Habitat*. Scottish Natural Heritage Commissioned Report 925.

A controlled study in 1991–1996 in a degraded fen meadow in the Netherlands (1) found that stripping topsoil reduced vegetation biomass after three months, but typically had no effect on vegetation cover after five years. After three months, above-ground biomass was significantly lower in a stripped area (20–240 g/m^2) than in an area that had not been stripped (200–490 g/m^2). After five years, both areas were dominated by velvety bentgrass *Agrostis canina* (36–37% cover) and contained the same three *Carex* sedge species (1–4% cover) but no *Sphagnum* moss. However, cover of purple moor grass *Molinia caerulea* was only 1% in the stripped area, compared to 21% in the unstripped area. Cover results were not tested for statistical significance. In 1991, 15–20 cm of topsoil was removed from 0.5 ha of degraded fen meadow. An adjacent area was not stripped. The meadow was historically drained but had been rewetted five years before stripping. Both areas were partially fertilized, partially limed and mown every August. In August 1994, above-ground vegetation was harvested in 60 x 60 cm quadrats (number not reported), then dried and weighed. Vegetation cover was estimated in 1996 (details not reported).

A study in 1991–1997 in a degraded fen meadow in Germany (2) reported that a plant community developed following topsoil removal, but its composition depended on the depth of soil removed. In plots with 40–60 cm of soil removed, the community contained wetland-characteristic herbs and tall rush species after six years. In plots with 20 cm of soil removed, species from drier grasslands were more abundant. All data were reported as a graphical analysis. The results were not tested for statistical significance. In February 1991, topsoil was removed from three 4,500 m^2 plots in a fen meadow historically used for agriculture. A different depth of soil was removed from each plot: 20, 40 or 60 cm. None of these plots were sown with hay. From 1992 to 1997, vegetation cover was estimated annually in five 4 m^2 quadrats/plot.

A replicated, randomized, paired, controlled study in 2000–2004 in a degraded peatland in Ohio, USA (3) found that plots stripped of topsoil contained significantly different plant communities to unstripped plots after four years, whilst plant species richness and diversity showed mixed results and biomass did not differ. Overall, the plant community in stripped plots contained more wetland-characteristic species and fewer upland-characteristic species than the community in unstripped plots (data reported as a graphical analysis). In one of two fields, stripped 25 m^2 plots contained more plant species than unstripped plots (24 vs 15) and were more diverse (data reported as a diversity index). In the other field, stripped plots contained fewer species than unstripped plots (13 vs 20) and were less diverse. Above-ground plant biomass did not differ between treatments in either field (stripped: 168–405; unstripped: 171–415 g/0.5 m^2). In 2000, twelve pairs of plots were established across two historically farmed peat fields. Topsoil (40–50 cm depth) was stripped from one plot in each pair but not from the other. Across the whole study area, the water table was raised and some seeds were sown (although the study states that most plants colonized naturally). In 2004, cover of every plant species was estimated in one 25 m^2 quadrat/plot. Above-ground biomass was collected from three 0.5 m^2 quadrats/plot, then dried and weighed.

A replicated, paired, controlled study in 2001–2005 in a degraded fen meadow in Germany (4) reported that topsoil removal increased the abundance of bog/fen characteristic plants. These results are not based on tests of statistical significance Five years following topsoil stripping, bog- and fen-characteristic plants occurred in up to 3% of quadrats with up to 3% cover/plot. Peatland-characteristic plants were not present in unstripped plots. In 2001, sixteen 6 x 6 m plots (in four blocks of four) were established in a drained, abandoned, nutrient-enriched fen meadow. Topsoil (30 cm depth) was stripped from eight plots but left on the others. None of these plots were sown with hay, but four stripped and four unstripped plots were fenced to exclude cattle. Between 2002 and 2005, vegetation cover was estimated in 16 permanent 1 m^2 quadrats/plot.

A replicated, paired, controlled study in 1991–2002 in a degraded fen meadow in the Netherlands (5) reported that plots stripped of topsoil contained different plant communities to unstripped plots. In particular, stripped plots were characterised by the absence of common haircap moss *Polytrichum commune* and star sedge *Carex echinata*. Plant communities in stripped plots also changed over time, whilst they remained stable in unstripped plots. All data were reported as a graphical analysis. The results were not tested for statistical significance. In 1991, surface vegetation and 10–15 cm of organic soil were stripped from two plots in an acidified fen meadow. Two adjacent plots were not stripped. Excess rainwater was drained by ditches. In 1993, 1997, 1999 and 2002, vegetation cover was estimated in representative areas of all four plots.

A replicated, paired, controlled study in 2006–2008 in a degraded, grassy fen in Northern Ireland, UK (6) found that plots stripped of surface peat had greater plant species richness than unstripped plots after two years, greater cover of rushes *Juncus* spp. and sedges *Carex* spp., but less cover of total vegetation and purple moor grass *Molinia caerulea*. Stripped plots contained more species/0.5 m^2 than unstripped plots (5.1 vs 3.9) and had greater cover of rushes (55 vs 2%) and sedges (13 vs <1%). However, stripped plots had less cover than unstripped plots of vegetation in total (65 vs 100%) and purple moor grass (11 vs 78%). Results were similar after one year, with the exception of species richness which did not differ significantly between stripped and unstripped plots (3.6 vs 3.9 species/0.5 m^2). In autumn 2006, four pairs of 5 x 5 m plots were established in a fen dominated by moor grass. Surface peat (15 cm depth) and vegetation were stripped from one plot in each pair, but not from the other. In July 2007 and October 2008, cover of every plant species was estimated in eight 70 x 70 cm quadrats/plot.

A replicated site comparison study in 2004–2007 in a drained fen meadow in Poland (7) reported that topsoil stripping changed the plant community composition, and that vascular plant cover and plant species richness increased over time after stripping. These results are not based on tests of statistical significance. Over three years following topsoil stripping, the overall plant community composition changed: it became less like degraded fen meadows, but also less like target fen meadow vegetation. The community also differed between plots stripped to different depths (data reported as a graphical analysis). Over the same time period, there were increases in vascular plant cover (from 2–3% to 58–75%) and plant species richness (from 5–8 species/4 m^2 to 18–19 species/4 m^2). In 2004, topsoil was stripped from eight 8 x 16 m plots in a drained fen meadow: 40 cm from four plots and 20 cm from the other four. All of these plots were left open to grazing by boar and deer, and were mown in 2006 and 2007. None of these plots were sown with hay. Vegetation cover and plant species were recorded annually between 2004 (after soil removal) and 2007, in each plot and in nearby degraded and target (reference) meadows.

A replicated, controlled study in 2008–2011 in a degraded fen in Poland (8) reported that plots stripped of topsoil developed a different plant community to unstripped and natural plots, but found that all plots contained a similar number of vascular plant species. After three years, the overall composition of the plant community differed between stripped, unstripped and natural plots. In particular, stripped plots had greater cover of jointleaf rush *Juncus articulatus*, cattail *Typha latifolia* and common duckweed *Lemna minor* than both unstripped and natural plots (data reported as a graphical analysis; differences not tested for statistical significance). However, the number of vascular plant species did not significantly differ between treatments (data not reported). In December 2008, 60 cm depth of topsoil was stripped from two 0.5 ha plots in the drained, degraded fen. Soil was not stripped from five adjacent plots. None of these plots were sown with hay. Ten plots in two natural fens were also monitored. In summer 2011, cover of every vascular plant species was estimated in each plot.

A replicated site comparison study in 2013 in six degraded rich fens in Belgium and the Netherlands (9) found that plots stripped of surface peat contained significantly less herb biomass than unstripped plots after 3–18 years, but had significantly greater bryophyte cover and plant species richness (data not reported). The higher species richness in stripped plots applied to both the total number of plant species and the number of fen-characteristic plant species. The differences in herb biomass and species richness were consistent in all eight sites. In June 2003, vegetation cover was recorded in eight 2 x 2 m quadrats/fen: four quadrats in an area stripped of surface peat and four in an unstripped area. In the stripped areas, 15–30 cm of peat had been removed (leaving some peat below) 3–18 years previously. Historically, all eight fens were drained and used for agriculture. Two had since been rewetted.

A controlled study in 1995–2008 in a degraded fen meadow in the Netherlands (10) reported that plots stripped of topsoil (and rewetted) developed different plant communities to unstripped (and drier) plots, with more plant species and greater moss/liverwort cover. Most of these results were not tested for statistical significance. Over 13 years, restored plots developed a different plant community (with fen meadow-characteristic species) to unrestored plots (dominated by species characteristic of drier, nutrient-rich sites; data reported as a graphical analysis). After 13 years, there were 24 species/4 m^2 in restored plots (vs 21 species/4 m^2 in unrestored plots). Restored plots also had significantly greater moss/liverwort cover (78–83%) than unrestored plots (23%). Results were similar in areas with shallow and deep topsoil removal. In 1995, an area of drained fen meadow was restored by stripping topsoil (shallow: 20cm; deep: 40 cm) and rewetting (by blocking local drainage ditches). The study does not distinguish between the effects of these interventions. An adjacent area was not stripped of topsoil or locally rewetted. Two interventions affected the whole site: additional rewetting by blocking a large drainage ditch in 2000, and reinstated annual mowing from 2001. Between 1997 and 2008, cover of every plant species was estimated in permanent 4 m^2 plots: 16 in the restored area and two in the unrestored area.

(1) van Duren I.C., Strykstra R.J., Grootjans A.P., ter Heerdt G.N.J. & Pegtel D.M. (1998) A multidisciplinary evaluation of restoration measures in a degraded *Cirsio-Molinietum* fen meadow. *Applied Vegetation Science*, 1, 115–130.

(2) Patzelt A., Wild U. & Pfadenhauer J. (2001) Restoration of wet fen meadows by topsoil removal: vegetation development and germination biology of fen species. *Restoration Ecology*, 9, 127–136.
(3) Hausman C.E., Fraser L.H., Kershner M.W. & de Szalay F.A. (2007) Plant community establishment in a restored wetland: effects of soil removal. *Applied Vegetation Science*, 10, 383–390.
(4) Rasran L., Vogt K. & Jensen K. (2007) Effects of topsoil removal, seed transfer with plant material and moderate grazing on restoration of riparian fen grasslands. *Applied Vegetation Science*, 10, 451–460.
(5) van der Hoek D. & Heijmans M.P.D. (2007) Effectiveness of turf stripping as a measure for restoring species-rich fen meadows in suboptimal hydrological conditions. *Restoration Ecology*, 15, 627–637.
(6) Reid N., McEvoy P.M. & Preston J.S. (2009) Efficacy of sod removal in regenerating fen vegetation for the conservation of the marsh fritillary butterfly *Euphydryas aurinia*, Montiaghs Moss Nature Reserve, County Antrim, Northern Ireland. *Conservation Evidence*, 6, 31–38.
(7) Klimkowska A., Kotowski W., van Diggelen R., Grootjans A.P., Dzierża P. & Brzezińska K. (2010) Vegetation re-development after fen meadow restoration by topsoil removal and hay transfer. *Restoration Ecology*, 18, 924–933.
(8) Hedberg P., Kozub Ł. & Kotowski W. (2014) Functional diversity analysis helps to identify filters affecting community assembly after fen restoration by top-soil removal and hay transfer. *Journal for Nature Conservation*, 22, 50–58.
(9) Emsens W.-J., Aggenbach C.J.S., Smolders A.J.P. & van Diggelen R. (2015) Topsoil removal in degraded rich fens: can we force an ecosystem reset? *Ecological Engineering*, 77, 223–232.
(10) Klimkowska A., van der Elst D.J.D. & Grootjans A.P. (2015) Understanding long-term effects of topsoil removal in peatlands: overcoming thresholds for fen meadow restoration. *Applied Vegetation Science*, 18, 110–120.

12.8 Bury upper layer of peat/soil

B Ⓕ S

- We captured no evidence for the effect of burying the upper layer of peat or soil (without planting afterwards) on peatland vegetation.

Background

Damaged peatlands may be covered by a layer of non-peat soil. Alternatively, the surface peat may contain excess nutrients or a seed bank of undesirable plants (e.g. left over from agricultural use), or may be too acidic (e.g. as a result of atmospheric deposition). Burying this upper layer (and any vegetation on it) under deeper peat layers, for instance by deep ploughing, could make these nutrients inaccessible to plants and prevent undesirable plants from growing. It will also create bare peat with spaces for plants to colonize (Glen *et al.* 2017). By replacing the surface layers under the deeper peat, the ground level could be maintained.

CAUTION: Soil burial may be unsuitable for wetter peatlands as heavy machinery involved may churn and compress the peat soil. Burying surface peat from bogs may expose fen peat, which has different chemical properties to bog peat and will not (in the short term) support bog vegetation (Lindsay & Clough 2016).

Related interventions: strip/remove upper layer of peat or soil (Section 12.7); disturb peatland surface (Section 12.9); bury upper layer of peat/soil before planting (Section 13.12).

Glen E., Price E.A.C., Caporn S.J.M., Carroll J.A., Jones L.M. & Scott R. (2017) Evaluation of topsoil inversion in UK habitat creation and restoration schemes. *Restoration Ecology*, 25, 72–81.

Lindsay R.A. & Clough J. (2016) *A Review of the Influence of Ombrotrophic Peat Depth on the Successful Restoration of Bog Habitat*. Scottish Natural Heritage Commissioned Report 925.

12.9 Disturb peatland surface to encourage growth of desirable plants

B Ⓕ S

- **Two studies** examined the effect of disturbing the peat surface (without planting) on peatland vegetation. Both studies were in fens.
- **Plant community composition (2 studies):** Two replicated, paired, controlled, before-and-after studies (one also randomized) in fens in Germany[1] and Sweden[2] reported that soil disturbance affected development of the plant community over 2–3 years. In Germany[1], disturbed plots developed greater cover of weedy species from the seed bank than undisturbed plots. In Sweden[2], the community in disturbed and undisturbed plots became less similar over time.
- **Characteristic plants (2 studies):** The same two studies reported that wetland- or fen-characteristic plant species colonized plots that had been disturbed (along with other interventions). The study in Germany[1] noted that peat-forming species did not colonize the fen.

Background

Mechanically disturbing the peat or soil surface (e.g. by tilling, ploughing or scarifying) can encourage the growth of desirable plants. It can create bare patches, clear of previous dominant species, in which peatland plants can grow. Peatland plants may colonize from nearby patches or germinate from seeds or spores already in the soil. Colonizing plants will typically be fast-growing, weedy species – but sometimes these are desirable members of natural peatland plant communities, or are desirable to provide some cover that can shelter and nurse other colonizing plants.

CAUTION: Disturbance can destroy the physical structure of the peat and aerate the peat more than usual. Deep ploughing could mix the peat with underlying mineral soil.

Related interventions: physically damage problematic plants, including by disturbing the peat/soil (Section 9.3); bury upper layer of peat or soil, as opposed to shallow disturbance only (Section 12.8).

A replicated, paired, controlled, before-and-after study in 1996–1998 in a degraded fen in Germany (1) reported that ploughed plots developed different plant communities to unploughed plots over two years. Specifically, ploughed plots developed greater cover than unploughed plots of weedy species from the seed bank such as toad rush *Juncus bufonius* and pale persicaria *Polygonum lapathifolium*. All plots were colonized by wetland-characteristic species such as cattail *Typha latifolia* and common rush *Juncus effusus*, but not sedges *Carex* spp. or common reed *Phragmites australis*. Data were reported as graphical analyses. Results were not tested for statistical significance. In 1996, two pairs of plots were established in a historically drained fen. In each pair, one plot was ploughed to a depth of 20 cm and one was not ploughed. Then, the surface of all plots was irrigated with lake water. Before intervention in 1996, then in 1997 and 1998, vegetation cover was estimated in a representative 16 m^2 area in each plot.

A replicated, randomized, paired, controlled, before-and-after study in 2002–2005 in two degraded rich fens in Sweden (2) reported that disturbing surface peat changed plant community composition and cover. The cover results were not tested for statistical significance. Disturbance significantly altered the development of the plant community over three years (data reported as a graphical analysis). Disturbed plots consistently had lower cover than undisturbed plots of *Sphagnum* mosses (disturbed: 0–2%; undisturbed; 2–25%) and purple moor grass *Molinia caerulea* (disturbed: 1–9%; undisturbed; 23–50%). Cover of common reed *Phragmites australis*, sedges *Carex* spp. and common cottongrass *Eriophorum angustifolium* showed mixed responses to disturbance amongst sites, species or other treatments applied to plots. Seventeen fen-characteristic plant species colonized disturbed plots (data not reported for undisturbed plots). In autumn 2002, sixty-four 2.5 x 2.5 m plots were established (in four blocks of 16) across two degraded fens. Thirty-two plots (eight random plots/block) were cleared of vegetation and dug over (top 10–20 cm of peat disturbed). The other plots were not disturbed. Additionally, trees had been removed from all plots and some plots were rewetted and/or mown. In 2002 (before intervention) and 2005, cover of every plant species was estimated in one 0.25 m^2 quadrat/plot.

(1) Richert M., Dietrich O., Koppsich D. & Roth S. (2000) The influence of rewetting on vegetation development and decomposition in a degraded fen. *Restoration Ecology*, 8, 186–195.
(2) Mälson K., Sundberg S. & Rydin H. (2010) Peat disturbance, mowing, and ditch blocking as tools in rich fen restoration. *Restoration Ecology*, 18, 469–478.

12.10 Add inorganic fertilizer

B F S

- **Three studies** examined the effect of adding inorganic fertilizer (without planting) on peatland vegetation. Two studies were in bogs[2,3] and one was in a fen meadow[1].
- **Vegetation cover (1 study):** One replicated, randomized, paired, controlled, before-and-after study in a bog in New Zealand[3] reported that fertilizing typically increased total vegetation cover.
- **Vegetation structure (1 study):** One replicated, paired, controlled study in a fen meadow in the Netherlands[1] found that fertilizing with phosphorous typically increased total above-ground vegetation biomass, but other chemicals typically had no effect.
- **Overall plant richness/diversity (1 study):** One replicated, randomized, paired, controlled, before-and-after study in a bog in New Zealand[3] reported that fertilizing typically increased plant species richness.
- **Growth (1 study):** One replicated, controlled, before-and-after study in a bog in Germany[2] found that fertilizing with phosphorous typically increased herb and shrub growth rate, but other chemicals had no effect.
- **Other (3 studies):** Three replicated, controlled studies in a fen meadow in Germany[1] and bogs in Germany[2] and New Zealand[3] reported that effects of fertilizer on peatland vegetation were more common when phosphorous fertilizer was added, than when nitrogen or potassium were added.

Background

Inorganic fertilizers could be used to manage nutrients in peatlands and speed up revegetation. Plant growth might be limited by a lack of nutrients overall, or of a specific nutrient, after drainage, mining, vegetation harvest or pollution. Commonly added nutrients include nitrogen (N), phosphorous (P) and/or potassium (K).

CAUTION: Some peatlands are characterized by low nutrient availability (Rydin & Jeglum 2013). Adding fertilizer might be a short-term solution to encourage initial plant growth, but could lead to undesirable long-term increases in nutrient levels.

Related intervention: add inorganic fertilizer to complement planting (Section 13.2).

Rydin H. & Jeglum J.K. (2013) *The Biology of Peatlands, Second Edition*. Oxford University Press, Oxford.

A replicated, paired, controlled study in 1994 in a degraded fen meadow in the Netherlands (1) found that adding fertilizer increased plant biomass in 10 of 24 comparisons. The other comparisons were non-significant increases. After three months, above-ground vegetation biomass was greater in plots fertilized with phosphorous (80–370 g/m^2) than in unfertilized plots (20–200 g/m^2). The same was true for plots fertilized with phosphorous and nitrogen (220–460 g/m^2) vs plots fertilized only with nitrogen (30–270 g/m^2), and for plots fertilized with potassium, phosphorous *and* nitrogen (240–490 g/m^2) vs plots fertilized with potassium and phosphorous *or* nitrogen (30–300 g/m^2). In May 1994, 1 m^2 plots (number not reported) were established in a rewetted fen meadow. Each plot received one fertilizer treatment: no fertilizer, N, P, K, N+P, N+K, P+K or N+P+K. Half of the plots were in an area stripped of topsoil. In August 1994, above-ground vegetation was harvested in a 60 x 60 cm quadrat in each plot, then dried and weighed.

A replicated, controlled, before-and-after study in 1995 in a historically mined raised bog in Germany (2) found that fertilizer increased seedling growth in 15 of 48 comparisons, all involving phosphorous, but had no effect in the other 33 comparisons. After four months, seedlings in plots fertilized with phosphorous (either alone or in combination with nitrogen and potassium) were significantly taller than seedlings in unfertilized plots in 15 of 24 comparisons (for which fertilized: 2–18 cm; unfertilized: 1–4 cm). Seedlings in plots fertilized only with nitrogen or potassium were never significantly taller than unfertilized seedlings (0 of 24 comparisons; fertilized: 1–5 cm; unfertilized: 2–4 cm). In spring 1995, six 16 m^2 plots of recently rewetted bare peat received each fertilizer treatment: N, P, K, or a mix of all three. Eight additional plots were not fertilized. After four months, all seedlings of six plant species (four herbs and two shrubs) were measured in every plot.

A replicated, randomized, paired, controlled, before-and-after study in 1998–2000 in a historically mined raised bog in New Zealand (3) reported that fertilized plots typically contained more plant species and had greater vegetation cover than unfertilized plots. These results are not based on tests of statistical significance. After two years, fertilized plots contained more plant species than unfertilized plots in 11 of 12 comparisons (fertilized: 3–8 species; unfertilized: 2–6 species). Fertilized plots had greater cover of two peatland-characteristic plants: manuka *Leptospermum scoparium* in 6 of 9 comparisons (for which fertilized: 1–92%; unfertilized: 0–87%) and bamboo rush *Sporadanthus ferrugineus* in 5 of 9 comparisons (for which fertilized: 2–27%; unfertilized: 1–8%). Total vegetation cover was also higher in fertilized plots in 6 of 9 comparisons. In March 1998, twenty-four plots (each 25 m^2) were established, in six blocks, on bare rewetted peat. Six plots (one random plot/block) received each fertilizer treatment: N, P, N+P, or none. None of these plots were sown. In June 2000, canopy cover was estimated for every plant species in each plot.

(1) van Duren I.C., Strykstra R.J., Grootjans A.P., ter Heerdt G.N.J. & Pegtel D.M. (1998) A multidisciplinary evaluation of restoration measures in a degraded *Cirsio-Molinietum* fen meadow. *Applied Vegetation Science*, 1, 115–130.
(2) Sliva J. & Pfadenhauer J. (1999) Restoration of cut-over raised bogs in southern Germany – a comparison of methods. *Applied Vegetation Science*, 2, 137–148.
(3) Schipper L.A., Clarkson B.R., Vojvodic-Cukovic M. & Webster R. (2002) Restoring cut-over restiad peat bogs: a factorial experiment of nutrients, seed and cultivation. *Ecological Engineering*, 19, 29–40.

12.11 Cover peatland with organic mulch

Ⓑ Ⓕ Ⓢ

- **Two studies** examined the effect on peatland vegetation of covering a peatland with organic mulch (without planting). Both studies were in bogs (but in one study[1], being restored as a fen).
- **Vegetation cover (2 studies):** One replicated, randomized, paired, controlled, before-and-after study in a bog in Canada[1] found that covering bare peat with straw mulch did not affect cover of fen-characteristic plants. One replicated, controlled, before-and-after study in a bog in Australia[2] reported that plots mulched with straw had similar *Sphagnum* moss cover to unmulched plots.
- **Characteristic plants (1 study):** One replicated, randomized, paired, controlled, before-and-after study in a bog in Canada[1] found that covering bare peat with straw mulch increased the number of fen characteristic plants present, but did not affect their cover.

Background

Peatland vegetation may be killed by hot, dry conditions on bare peat surfaces (e.g. Sagot & Rochefort 1996). Organic mulches (e.g. straw, grass cuttings or shrub roots) can be placed on the peatland surface to stabilize temperatures and humidity, and provide shade. This may create a more hospitable environment for establishment and growth of peatland vegetation (Good *et al.* 2009). Typically, mulch is applied sparsely enough that some light can still reach the peat surface. CAUTION: Mulches may contain seeds of undesirable plants. Sterilization before application can kill these.

This section considers the effect of adding mulch without adding living vegetation. The mulch is intended to help existing vegetation e.g. remnant moss patches, or seedlings that germinate from seeds already in the peat.

Related interventions: cover peat with something other than mulch e.g. plastic sheets or shade cloths (Section 12.12); add mulch after planting (Section 13.4).

Good R., Wright G., Whinam J. & Hope G. (2009) Restoration of mires of the Australian Alps following the 2003 wildfires. Pages 353–362 in: S.G. Haberle, J. Stevenson & M. Prebble (eds.) *Altered Ecologies: Fire, Climate and Human Influence on Terrestrial Landscapes*. Terra Australis 32, Australian National University e-press, Canberra, Australia.

Sagot C. & Rochefort L. (1996) Tolérance des sphaignes à la dessiccation (Tolerance of *Sphagnum* mosses to desiccation; in French). *Cryptogamie, Bryology-Lichénologie*, 17, 171–183.

A replicated, randomized, paired, controlled, before-and-after study in 2001–2002 in a historically mined bog in Quebec, Canada (1) found that mulching with straw increased the number of fen-characteristic plant species but had no effect on fen-characteristic plant cover. Note that the aim of this study was to create a fen, as the post-mining peat chemistry was more like a fen than a bog. Before sowing, no vegetation was present. After two growing seasons, there were more plant species typical of local fens in mulched plots (8 species) than unmulched plots (5 species). Fen plant cover did not significantly differ between mulched (6%) and unmulched plots (10%). In spring 2001, eighteen 5 x 5 m plots were established, in three blocks of six. Nine plots (three random plots/block) were mulched with straw (1,500 kg/ha). The other plots were not mulched. All plots had previously been rewetted, raked and fertilised. None of these plots were sown. In August 2002, cover of every plant species was estimated in ten 30 x 30 cm quadrats/plot.

A replicated, controlled, before-and-after study in 2003–2007 in a fire-damaged bog in Australia (2) reported that mulching with straw had no effect on *Sphagnum* moss cover. This result is not based on a test of statistical significance. After 40 months, *Sphagnum* cover was similar in straw-mulched (8.6%) and unmulched plots (7.8%). This followed fluctuations over the 40 months, when *Sphagnum* cover was sometimes higher in mulched than unmulched plots but sometimes lower. Immediately before shading, plots had approximately 3% *Sphagnum* cover. In January 2003, the focal bog was burned by a wild fire. In October 2003, five burned plots (3 x 15 m) were mulched with sterilized straw (2 tonnes/ha). Five additional plots were not mulched. Vegetation cover was recorded in 0.25 m^2 quadrats: five across the bog in October 2003, then one/plot every six months until March 2007.

(1) Cobbaert D., Rochefort L. & Price J.S. (2004) Experimental restoration of a fen plant community after peat mining. *Applied Vegetation Science*, 7, 209–220.
(2) Whinam J., Hope G., Good R. & Wright G. (2010) Post-fire experimental trials of vegetation restoration techniques in the peatlands of Namadgi (ACT) and Kosciuszko National Parks (NSW), Australia. Pages 363–379 in: S.G. Haberle, J. Stevenson & M. Prebble (eds.) *Altered ecologies: fire, climate and human influence on terrestrial landscapes*. Terra Australis 32, Australian National University e-press, Canberra, Australia.

12.12 Cover peatland with something other than mulch Ⓑ Ⓕ Ⓢ

- **Two studies** examined the effect on peatland vegetation of covering a peatland with something other than mulch (without planting). Both studies were in bogs.
- **Vegetation cover (2 studies):** One replicated, controlled, before-and-after study in a bog in Germany[1] reported that covering bare peat with fleece or fibre mats did not affect the number of seedlings of five herb/shrub species. One replicated, controlled, before-and-after study in bogs in Australia[2] reported that recently-burned plots shaded with plastic mesh developed greater cover of native plants, forbs and *Sphagnum* moss than unshaded plots.

Background

Peatland vegetation may be killed by hot, dry and bright conditions on bare peat surfaces (e.g. Harley *et al.* 1989; Sagot & Rochefort 1996). Covers (e.g. plastic sheets, fleece or fibre mats) can maintain more stable temperatures and humidity, and offer some shading. This may create a more hospitable environment for establishment and growth of peatland vegetation (Good *et al.* 2009). The precise effect (mainly affecting light and/or moisture) depends on the material and its height above the peatland.

This section considers the effect of adding covers without adding vegetation. The cover is intended to help existing vegetation e.g. remnant moss patches, or seedlings that germinate from seeds already in the peat. We use the term *mesh* to describe covers such as shade cloths, gauze and netting that are used to shade a peatland.

Related interventions: cover peat with organic mulch (Section 12.11); add cover other than mulch after planting (Section 13.5).

Good R., Wright G., Whinam J. & Hope G. (2009) Restoration of mires of the Australian Alps following the 2003 wildfires. Pages 353–362 in: S.G. Haberle, J. Stevenson & M. Prebble (eds.) *Altered Ecologies: Fire, Climate and Human Influence on Terrestrial Landscapes*. Terra Australis 32, Australian National University e-press, Canberra, Australia.

Harley P.C., Tenhunen J.D., Murray K.J. & Beyers J. (1989) Irradiance and temperature effects on photosynthesis of tussock tundra *Sphagnum* mosses from the foothills of the Philip Smith Mountains, Alaska. *Oecologia*, 79, 251–259.

Sagot C. & Rochefort L. (1996) Tolérance des sphaignes à la dessiccation (Tolerance of *Sphagnum* mosses to desiccation; in French). *Crytogamie, Bryology-Lichénologie*, 17, 171–183.

A replicated, controlled, before-and-after study in 1993–1995 in a historically mined raised bog in Germany (1) reported that covering plots with fleece or fibre mat did not affect seedling numbers for five plant species. These results were not tested for statistical significance. After 1–2 years, covered and uncovered plots contained a similar number of seedlings. There were 3 seedlings/400 cm^2 for purple moor grass *Molinia caerulea*. There was <1 seedling/400 cm^2 for four other species: beaked sedge *Carex rostrata*, common cottongrass *Eriophorum angustifolium*, sheathed cottongrass *Eriophorum vaginatum* and heather *Calluna vulgaris*. In autumn 1993, fifteen 1 m^2 plots were established on bare rewetted peat (mined until 1986). Five plots were covered with synthetic fleece, five were covered with wide-meshed jute fibre mat and five were not covered. No seeds were added to these plots. Covers were removed and seedlings counted in summer 1994 (two plots/treatment) and 1995 (three plots/treatment).

A replicated, controlled, before-and-after study in 2003–2007 in two fire-damaged bogs in Australia (2) found that plots shaded with plastic mesh developed greater vegetation cover than unshaded plots. After 40 months, shaded plots had significantly greater cover of native plants in general, and of forbs, than unshaded plots (data not reported). *Sphagnum* moss cover was 10% in shaded plots compared to 8% in unshaded plots (difference not tested for statistical significance). Immediately before shading, plots had 3% *Sphagnum* cover on average. In January 2003, the focal bogs were burned by a wild fire. In October 2003, ten burned plots (3 x 15 m; five plots/bog) were shaded with plastic mesh (blocking 70% of incoming light). Fifteen additional plots were left uncovered. Vegetation cover was recorded in 0.25 m^2 quadrats: five/bog in October 2003, and one/plot in March 2007.

(1) Sliva J. & Pfadenhauer J. (1999) Restoration of cut-over raised bogs in southern Germany – a comparison of methods. *Applied Vegetation Science*, 2, 137–148.
(2) Whinam J., Hope G., Good R. & Wright G. (2010) Post-fire experimental trials of vegetation restoration techniques in the peatlands of Namadgi (ACT) and Kosciuszko National Parks (NSW), Australia. Pages 363–379 in: S.G. Haberle, J. Stevenson & M. Prebble (eds.) *Altered ecologies: fire, climate and human influence on terrestrial landscapes*. Terra Australis 32, Australian National University e-press, Canberra, Australia.

12.13 Stabilize peatland surface to help plants colonize Ⓑ Ⓕ S

- **One study** examined the effect of stabilizing the peatland surface (without planting) on peatland vegetation. The study was in a bog.
- **Vegetation cover (1 study):** One controlled, before-and-after study in a bog in the UK[1] found that pegging coconut fibre rolls onto almost-bare peat did not affect the development of vegetation cover (total, mosses, shrubs or cottongrasses).

Background

Peatland plants may struggle to colonize loose bare peat. Seeds, spores or young plants can be washed or blown away. The peatland surface could be stabilized by pegging fibre netting (e.g. geojute) into the peat or applying fibre rolls to act as wind/water breaks. If made of organic material, these stabilizing aids should degrade once peatland vegetation has colonized.

Related interventions: cover peat with mulch or other material, which may stabilize the peat surface as well as controlling the microclimate (Sections 12.11 and 12.12).

A controlled, before-and-after study in 2007–2010 in a degraded blanket bog in England, UK (1) found that adding coconut fibre rolls to stabilize the peat surface had no effect on vegetation cover. Comparing data from before intervention and three years after, vegetation cover increased by a similar amount in areas with and without the rolls. This was true for total vegetation cover (with rolls: from 6 to 10%; without: from 15 to 20%), moss cover (with rolls: from 0 to 1.0%; without: from 0 to 2.5%), dwarf shrub cover (with rolls: from 0.5 to 1%; without: from 0.5 to 4%) and common cottongrass *Eriophorum angustifolium* cover (with rolls: from 1 to 3%; without: from 4 to 7%). In March 2007, coconut fibre rolls were pegged onto an area of almost-bare peat to stabilize it. An adjacent area was left untreated. Sheep were excluded from both areas before the study began. In 2007 (before intervention) and 2010, vegetation cover was estimated in thirty 2 x 2 m quadrats/area.

(1) Anderson P., Worrall P., Ross S., Hammond G. & Keen A. (2011) *United Utilities Sustainable Catchment Management Programme Volume 3: The Restoration of Highly Degraded Blanket Bog*. Penny Anderson Associates Project Report.

12.14 Introduce nurse plants

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect of introducing nurse plants on naturally colonizing, focal peatland vegetation.

Background

Nurse or companion plants can be planted to help naturally recolonizing peatland vegetation (Padilla & Pugnaire 2006). Nurse plants can bind together loose peat and reduce harsh environmental conditions (temperature fluctuations, desiccation and strong sunlight). Clearly, nurse plants that can tolerate these conditions must be selected. Invasive species (that spread easily) and species that may outcompete focal plants (physically or chemically) should be avoided. Instead, it is expected that nurse plants are eventually outcompeted *by* focal plants.

Herbs are common nurse plants in temperate peatlands, creating shelter for mosses. They may be directly planted or added as seed (the latter being a more efficient way to cover larger areas). Haircap moss *Polytrichum strictum* may act as a nurse plant for *Sphagnum* mosses: a correlative study in Canada found that natural recolonization of *Sphagnum* only occurred in carpets of haircap moss (Groeneveld *et al*. 2007). In tropical peat swamps, light-tolerant trees can be used to shelter shade-loving trees.

Related interventions: restore or create a peatland using multiple interventions, which commonly includes introducing nurse plants (Section 12.1); introduce nurse plants before planting focal peatland plants (Section 13.6).

Groeneveld E.V.G., Masse A. & Rochefort L. (2007) *Polytrichum strictum* as a nurse-plant in peatland restoration. *Restoration Ecology*, 15, 709–719.

Padilla F.M. & Pugnaire F.I. (2006) The role of nurse plants in the restoration of degraded environments. *Frontiers in Ecology and the Environment*, 4, 196–202.

12.15 Build artificial bird perches to encourage seed dispersal

B F Ⓢ

- **One study** examined the effect on peatland vegetation of building artificial bird perches. The study was in a tropical peat swamp.
- **Vegetation cover (1 study):** One replicated, paired, controlled study in a peat swamp forest in Indonesia[1] found that artificial bird perches had no significant effect on seedling abundance.

Background

Artificial bird perches may help to restore peatland forests. Perches are placed near to remaining forest patches to encourage birds to fly out, perch and defecate onto degraded land. The seeds transported in this way might belong to peatland plants (especially tropical swamp forest trees) that would otherwise not disperse into degraded land. Consequently, the abundance and diversity of seedlings may increase in areas with perches (Agra *et al.* 2016).

Related intervention: directly introduce seeds or vegetation fragments containing seeds (Section 12.18).

Agra H., Carmel Y., Smith R.K. & Ne'eman G. (2016) *Forest Conservation: Global Evidence for the Effects of Interventions*. University of Cambridge, Cambridge, UK.

A replicated, paired, controlled study in 2007–2008 in a degraded, burned, peat swamp forest in Kalimantan, Indonesia (1) found that installing artificial bird perches along a forest edge had no effect on tree seedling abundance. After one year, seedling density was not significantly different under artificial perches (2–6 seedlings/m^2) and in adjacent plots not under perches (1–2 seedlings/m^2). Most seedlings under the perches were tampohot *Syzygium* sp. (79% of all seedlings). In July 2007, ten 8 m tall artificial bird perches were erected in logged and burned peatland, 50 to 200 m from the edge of a remnant forest patch. Two 1 m^2 plots were monitored for each perch: one directly underneath the perch and one next to it. All seedlings initially present were marked. In July 2008, new tree seedlings were counted.

(1) Graham L.L.B. & Page S.E. (2011) Artificial bird perches for the regeneration of degraded tropical peat swamp forest: a restoration tool with limited potential. *Restoration Ecology*, 20, 631–637.

Interventions: Introduce peatland vegetation

12.16 Directly plant whole peatland plants

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Background

This section considers introducing peatland vegetation by planting whole plants i.e. placing individual seedlings, mature plants, shoots, cuttings, rhizomes or sods directly into peat/soil. Plants may be collected from natural peatlands or grown in greenhouses/laboratories. Direct introduction of peatland vegetation might be necessary in severely degraded or bare peatlands. Natural revegetation (from remnant plants, seed banks or dispersal) might not happen, might be very slow or might not produce the desired mix of species.

CAUTION: Collecting vegetation from natural peatlands damages the donor site, although rapid recovery has been reported (Rochefort & Campeau 2002). Trees and shrubs are not natural features of all peatlands: growth of woody plants is a threat to many bogs and fens. Trees could dry out peatlands by taking up water.

Related interventions: spread vegetation onto peatland surface (Section 12.17); introduce seeds or vegetation fragments containing seeds (Section 12.18); restoration using more than three interventions, sometimes including planting (Sections 12.1 and 12.2); supporting interventions from this section, e.g. mulching or fertilizing, used without introducing vegetation (Sections 12.3–12.14); experimental tests of interventions to complement planting (Chapter 13).

Kooijman A.M., Beltman B. & Westhoff V. (1994) Extinction and reintroduction of the bryophyte *Scorpidium scorpioides* in a rich-fen spring site in the Netherlands. *Biological Conservation*, 69, 87–96.

Rochefort L. & Campeau S. (2002) Recovery of donor sites used for peatland restoration. Pages 244–251 in: G. Schmilewski & L. Rochefort (eds.) *IPS Symposium Proceedings: Peat in horticulture – quality and environmental challenges*. International Peat Society, Jyväskylä, Finland.

Sliva J. & Pfadenhauer J. (1999) Restoration of cut-over raised bogs in southern Germany – a comparison of methods. *Applied Vegetation Science*, 2, 137–148.

12.16.1 Directly plant peatland mosses

B F S

- **Seven studies** examined the effect on peatland vegetation of planting mosses. Six studies were in bogs[2,3,4,5,6,7] and one was in a fen[1].
- **Survival (1 study):** One study in Lithuania[7] reported that of 50 *Sphagnum*-dominated sods planted into a rewetted bog, 47 survived for one year.
- **Growth (2 studies):** Two before-and-after studies in a fen in the Netherlands[1] and bog pools in the UK[2] reported that mosses grew after planting.
- **Moss cover (5 studies):** Five before-and-after studies in a fen in the Netherlands[1] and bogs in Germany[3], Ireland[4,5], Estonia[5] and Australia[6] reported that after planting mosses, the area covered by moss increased in at least some cases. The study in the Netherlands[1] reported spread of planted moss beyond the introduction site. The study in Australia[6] was also controlled and reported that planted plots developed greater *Sphagnum* moss cover than unplanted plots.

A before-and-after study in 1989–1992 in a fen in the Netherlands (1) reported that transplanted shoots of scorpion moss *Scorpidium scorpioides* grew in length and spread to new parts of the fen. No statistical tests were carried out. Twenty months after transplant, "many" shoots had died but remaining shoots had grown 3 cm on average. New plants were found in 25 grid cells up to 1.2 m from original transplants. After three years, this had increased to plants in 80 grid cells up to 2 m from the transplants. In November 1989, five rings of live scorpion moss (3.5 cm diameter) were cut from an Irish fen and planted in the Dutch fen, where scorpion moss was absent. Five plants in each ring were marked 3 cm below the shoot tip. In July 1991, measurements were taken of shoot length (above the marks) and expansion of moss plants into a grid of 10 x 10 cm squares around the transplants. Expansion measurements were repeated in December 1992.

A before-and-after study in 1991 in a historically mined raised bog in England, UK (2) reported that planted *Sphagnum* moss grew within bog pools. Over the first 20 weeks after planting, feathery bog moss *Sphagnum cuspidatum* plants had grown by 10–15% per week. Recurved bog moss *Sphagnum recurvum* plants had grown by 6–

13% per week. Growth of both species was affected by liming (see Section 13.1) and fertilization (see Section 13.2). In 1991, individual *Sphagnum* plants (cut to 5 cm length) were submerged (30 cm deep) in 4 m^3 pools dug in the bog (number of plants and pools not reported). After 10 days, some pools were limed, fertilized or limed and fertilized. After 20 weeks, the length of all plants was measured.

A replicated before-and-after study in 1991–1995 in a historically mined raised bog in Germany (3) reported that transplanted sods of *Sphagnum* moss grew larger in one of five sites but did not grow (or shrunk) in the other four. No statistical tests were carried out. Sods of three *Sphagnum* species increased in diameter when planted at a site with sedge *Carex* sp. present (from 20 cm to 54–82 cm over four years). The species were Magellanic bog moss *Sphagnum magellanicum*, feathery bog moss *Sphagnum cuspidatum* and red bog moss *Sphagnum capillifolium*. All three species did not grow, or shrunk, when planted between *Eriophorum* cottongrass at three sites or into an unvegetated site (from 20 cm to 0–23 cm). Cover of living *Sphagnum* within the sods showed similar responses. In 1991, five sites in a historically mined but rewetted bog were planted with 20 sods (25 cm thick, 12 cm diameter) of each *Sphagnum* species. From 1992 to 1995, sod diameter and cover of living *Sphagnum* were recorded.

A replicated before-and-after study in 1998–2001 in a bog in Ireland (4) reported that transplanted sods of *Sphagnum* moss grew in bare or moss-covered peat. No statistical tests were carried out. Two years after transplantation to a soaked bare peat surface, sods of transplanted *Sphagnum* sods covered 1,350–1,400 cm^2 (compared to 480–510 cm^2 when planted). Similarly, two years after transplantation into established feathery bog moss *Sphagnum cuspidatum*, transplanted *Sphagnum* sods covered 1,290–1,710 cm^2 (compared to 350–510 cm^2 when planted). In 1998 or 1999, sods of Magellanic bog moss *Sphagnum magellanicum* and papillose bog moss *Sphagnum papillosum* were cut from existing bogs. Three sods of each species, approximately 500 cm^2 and 10 cm deep, were transplanted to depressions: bare or covered with feathery bog moss. Sod surface areas were measured annually.

A replicated, paired, before-and-after study in 2003–2006 in two raised bogs in Ireland and Estonia (5) found that transplants of *Sphagnum* moss survived for three years at 5–125% of their original size. Three species were transplanted. For two species (red bog moss *Sphagnum rubellum* and rusty bog moss *Sphagnum fuscum*), larger 14 cm diameter transplants grew, or shrunk less (84–127% original size) than smaller 7 cm diameter transplants (25–113% original size). For the other species (feathery bog moss *Sphagnum cuspidatum*), shrinkage was not significantly affected by transplant size (large 18–56%; small 5–50% original size). In June 2003, 5–6 large (14 cm diameter) and 20–24 small (7 cm diameter) cores of single moss species, each 20 cm thick, were transplanted to bogs dominated by Magellan's bog moss *Sphagnum magellanicum*. Transplants were arranged in sets of one large with four small. Fragment areas were measured from photographs taken in September 2006.

A replicated, controlled, before-and-after study in 2003–2007 in seven burned bogs in Australia (6) reported that plots planted with *Sphagnum* moss developed greater *Sphagnum* cover than unplanted plots, especially when shaded. These results were not tested for statistical significance. Immediately before intervention, *Sphagnum* cover was approximately 3%. After 40 months, plots planted with *Sphagnum* sods had developed 9–21% *Sphagnum* cover: 9% if mulched with straw, 11% if shaded with a vertical cloth and 21% if shaded with a horizontal cloth. In comparison, unplanted plots had developed 8–10% *Sphagnum* cover: 8% with no intervention and 10% if shaded with a horizontal cloth. In October 2003, 75 plots were established across bogs recently burned by wild fire. In one bog, fifteen 45 m^2 plots were planted with sods of mixed *Sphagnum* species (30 cm thick, 400 cm^2). All sods were fertilized. Five planted plots then received each cover treatment: straw mulch, vertical shade cloth or horizontal shade cloth. In the same bog, five additional plots were covered with shade cloth but not planted. The remaining 55 plots across all seven bogs received no intervention. In October 2003 and 2007, *Sphagnum* cover was estimated in 5–20 quadrats/plot or bog. Quadrats were 0.25 m^2.

A study in 2006–2012 in a historically mined raised bog in Lithuania (7) reported that 94% of planted *Sphagnum*-dominated sods survived for one year. The study also reported that *Sphagnum* had started to grow on adjacent bare peat, but this was not quantified. In September 2011, 50 sods cut from a donor bog were transplanted to a degraded but rewetted bog. Each sod was 40 x 40 cm in area and 5–7 cm thick. The donor bog was dominated by rusty bog moss *Sphagnum fuscum*, red bog moss *Sphagnum capillifolium* and Magellan's bog moss *Sphagnum magellanicum* but the sods also contained vascular plants. The degraded bog had been rewetted by building dams and installing underground plastic membranes. Sod survival was recorded in 2012.

(1) Kooijman A.M., Beltman B. & Westhoff V. (1994) Extinction and reintroduction of the bryophyte *Scorpidium scorpioides* in a rich-fen spring site in the Netherlands. *Biological Conservation*, 69, 87–96.

(2) Money R.P. (1995) Re-establishment of a *Sphagnum* dominated flora on cut-over lowland raised bogs. Pages 405–422 in B.D. Wheeler, S.C. Shaw, W.J. Fojt & R.A. Robertson (eds.): *Restoration of Temperate Wetlands.* John Wiley and Sons Ltd., Chichester.

(3) Sliva J. & Pfadenhauer J. (1999) Restoration of cut-over raised bogs in southern Germany – a comparison of methods. *Applied Vegetation Science*, 2, 137–148.

(4) Smolders A.J.P., Tomassen H.B.M., van Mullekom L.P.M. & Roelofs J.G.M. (2003) Mechanisms involved in the re-establishment of *Sphagnum*-dominated vegetation in rewetted bog remnants. *Wetlands Ecology and Management*, 11, 403–418.

(5) Robroek B.J.M., van Ruijven J., Schouten M.G.C., Breeuwer A., Crushell P.H., Berendse F. & Limpens J. (2009) *Sphagnum* reintroduction in degraded peatlands: the effects of aggregation, species identity and water table. *Basic and Applied Ecology*, 10, 697–706.

(6) Whinam J., Hope G., Good R. & Wright G. (2010) Post-fire experimental trials of vegetation restoration techniques in the peatlands of Namadgi (ACT) and Kosciuszko National Parks (NSW), Australia. Pages 363–379 in: S.G. Haberle, J. Stevenson & M. Prebble (eds.) *Altered ecologies: fire, climate and human influence on terrestrial landscapes*. Terra Australis 32, Australian National University e-press, Canberra, Australia.

(7) Jarašius L., Pakalnis R., Sendžikaitė J. & Matelevičiūtė D. (2013) Experiments with restoration of raised bog vegetation in Aukštumala Raised Bog in Lithuania. Pages 225–229 in: M. Pakalne & L. Strazdiņa (eds.) *Raised Bog Management for Biological Diversity Conservation in Latvia*. University of Latvia, Riga.

12.16.2 Directly plant peatland herbs

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- **Five studies** examined the effect on peatland vegetation of planting herbaceous plants. Three studies were in fens or fen meadows[1,3,5] and two were in bogs[2,4].
- **Survival (3 studies):** Three replicated studies in a fen meadow in the Netherlands[1] and fens in the USA[3,5] reported that planted herbs survived over 2–3 years. However, for six of nine species only a minority of individuals survived.
- **Growth (2 studies):** Two replicated before-and-after studies in a bog in Germany[2] and fens in the USA[5] reported that individual planted herbs grew.

- **Vegetation cover (1 study):** One replicated, controlled, before-and-after study in Canada[4] found that planting herbs had no effect on moss, herb or shrub cover in created bog pools relative to natural colonization.

A replicated study in 1994–1995 in a degraded fen meadow in the Netherlands (1) reported that most planted herbs survived over one growing season, but after two growing seasons survival was lower and more variable. These results were not tested for statistical significance. After one growing season, 92–100% of planted carnation sedge *Carex panicea*, tawny sedge *Carex hostiana* and meadow thistle *Cirsium dissectum* had survived. After two growing seasons, survival was 8–88% across all species, but higher for carnation sedge (72–88%) than tawny sedge (8–20%) or meadow thistle (8–32%). For tawny sedge and meadow thistle, survival was lower in limed plots (8–20%) than unlimed plots (15–32%), and in topsoil-stripped plots (8–20%) than unstripped plots (8–32%). In May 1994, twenty 1 m^2 plots were each planted with 15 plants (five of each species). Five plots had been stripped and limed, five stripped but not limed, five limed but not stripped, and five neither stripped nor limed. All plots had been rewetted and were mown every August. In August 1994 and 1995, survival of all plants was recorded.

A replicated before-and-after study in 1991–1995 in a historically mined raised bog in Germany (2) reported that planted herbs grew. Three species were planted. After four years, sedge *Carex rostrata* plants had 40 shoots (vs 1 shoot when planted), common cottongrass *Eriophorum angustifolium* plants had 2 shoots (vs 1 shoot when planted) and tussocks of sheathed cottonsedge *Eriophorum vaginatum* were 70 cm in diameter (vs 40 cm after two years). These results were not tested for statistical significance. For the first two species, additional fertilized plants developed more shoots than unfertilized plants (sedges: 142 shoots/plant; common cottongrass: 6 shoots/plant) but fertilizer had no significant effect on sheathed cottonsedge tussock diameter. In 1991, twelve 3 x 35 m plots of bare rewetted peat were planted with the shoots and tussocks (one plant/3 m^2). In 1994, six plots were fertilized (nitrogen-phosphorous-potassium; 100 g/m^2). Plants were measured in 1991, 1993 and 1995.

A replicated study in 1992–1994 in a historically mined fen in Colorado, USA (3) reported that 7–65% of planted herbs survived over two years. Seven species were planted. Survival was 50% for water sedge *Carex aquatilis* seedlings. New clones had also appeared. Survival was 42–65% for rhizomes of three *Carex* sedge species, 26% for elk sedge *Kobresia simpliciuscula* rhizomes, 13% for common cottongrass *Eriophorum angustifolium* rhizomes and 7% for arctic rush *Juncus articus* rhizomes. Survival of some species was affected by water table depth. In June 1992, each species was planted into 10 or 27 separate 50 x 50 cm plots (10 plants/plot). Plots contained shallow surface peat (a "few" centimetres), had variable water levels and were cleared of existing vegetation. Rhizomes, supporting at least two live shoots, were transplanted immediately after collection. Water sedge seedlings were grown from seed in spring 1992. Survival was recorded in August 1994.

A replicated, controlled, before-and-after study in 1999–2003 in a historically mined bog in eastern Canada (4) found that planting in and around created pools did not significantly affect vegetation cover. After four years, planted and unplanted pools had similar cover of *Sphagnum* moss (13 vs 9%), other mosses (3 vs 3%), herbs (3 vs 5%), shrubs (6 vs 5%). In 1999, eight 6 x 8.5 m pools were created by excavating and rewetting a bog (blocking ditches and building embankments). In 2000, four pools were planted with four herb species. *Sphagnum* moss was also introduced to the water column. The study does not distinguish between the effects of planting herbs and mosses. The other four pools were not planted, although bog vegetation fragments were spread onto the rest of the peatland. In 2003, vegetation cover was recorded in 36 quadrats/pool, each 30 x 30 cm, along six bank-to-bank transects.

A replicated before-and-after study in 2007–2010 in three degraded fens in Colorado, USA (5) reported that 35–55% of transplanted water sedge *Carex aquatilis* survived over three years, and that surviving plants had grown. On average, sedge plants had more stems after three years (11 stems/plant) than when planted (2 stems/plant). This result was not tested for statistical significance. Mulching planted sedges significantly increased their survival (mulched: 55%; unmulched: 35%) and growth (mulched: 3–9 stems/plant; unmulched: 1–9 stems/plant). In July 2007, sedges were transplanted into 36 bare peat plots (12 plots/fen). Transplants were rhizomes with stems, dug from natural vegetation. Nine sedges were planted in each plot, approximately 35 cm apart. Eighteen plots (6 plots/fen) were also mulched with straw (immediately) and shredded aspen (after one year). In summer 2010, sedge survival and number of stems were counted.

(1) van Duren I.C., Strykstra R.J., Grootjans A.P., ter Heerdt G.N.J. & Pegtel D.M. (1998) A multidisciplinary evaluation of restoration measures in a degraded *Cirsio-Molinietum* fen meadow. *Applied Vegetation Science*, 1, 115–130.
(2) Sliva J. & Pfadenhauer J. (1999) Restoration of cut-over raised bogs in southern Germany – a comparison of methods. *Applied Vegetation Science*, 2, 137–148.
(3) Cooper D.J. & MacDonald L.H. (2000) Restoring the vegetation of mined peatlands in the southern Rocky Mountains of Colorado, USA. *Restoration Ecology*, 8, 103–111.
(4) Mazerolle M.J., Poulin M., Lavoie C., Rochefort L., Desrochers A. & Drolet B. (2006) Animal and vegetation patterns in natural and man-made bog pools: implications for restoration. *Freshwater Biology*, 51, 333–350.
(5) Chimner R.A. (2011) Restoring sedges and mosses into frost heaving iron fens, San Juan Mountains, Colorado. *Mires and Peat*, 8, Article 7.

12.16.3 Directly plant peatland trees/shrubs B F S

- **Eleven studies** examined the effect, on peatland vegetation, of planting trees/shrubs to restore or create forested/shrubby peatland.. Seven studies were in tropical peat swamps[2,3,6,7,9,10,11], three in bogs[4,5,8] and one in a fen[1].
- **Survival (10 studies):** Eight studies (seven replicated) in peat swamp forests in Thailand[2], Malaysia[3,10] and Indonesia[7,9] and bogs in Canada[4,5,8] reported that the majority of planted trees/shrubs survived over periods between 10 weeks and 13 years. Species with <50% survival included *Dacryodes* sp.[2], poplar[4] and katok[10]. One replicated study in a fen in the USA[1] reported that most planted willow cuttings died within two years. One study in a peat swamp forest in Indonesia[6] reported <5% survival of planted trees after five months, following unusually deep flooding.
- **Growth (5 studies):** Four studies (including two replicated, before-and-after) in peat swamp forests in Thailand[2], Indonesia[9,11] and Malaysia[10] reported that planted trees grew. One replicated before-and-after study in bogs in Canada[8] reported that planted shrubs grew.

A replicated study in 1992–1994 in a historically mined fen in Colorado, USA (1) reported that 12–33% of planted willow *Salix* spp. cuttings survived over two years. Four species were planted. Survival of myrtle-leaf willow *Salix myrtillifolia* was 33%, mountain willow *Salix monticola* 26%, hoary willow *Salix candida* 13% and barren ground willow *Salix brachycarpa* 12%. In June 1992, four plots (myrtle-leaf willow) or 27 plots (all other species) were planted with 10–20 fresh cuttings of each species. Plots were 0.5 x 0.5 m, contained a "few" centimetres of surface peat, had variable water levels and had been cleared of existing vegetation. Cuttings were woody stems approximately 30 cm long. Approximately 20 cm was buried below the soil surface. Half the leaves were removed prior to planting. Survival was recorded in August 1994.

A study in 1988–1997 in a degraded peat swamp in Thailand (2) reported that 22–97% of planted trees survived for four years or more, and that surviving trees grew. No statistical tests were carried out. For 22 of 28 planted species, at least 50% of planted trees survived for at least four years. Survival was highest for milkwood *Alstonia spathulata* and *Ixora grandifolia* (97% after four years) and lowest for *Dacryodes* sp. (22% after four years). For 13 species, survival rates over nine years were also reported and were similar to those after five and half years (within 2%). For 15 species, growth rates were reported. These species all grew, from 35–120 cm tall one year after planting to 110–340 cm tall four years after planting. Trees (number not reported) were planted into the degraded peat swamp in 1988 (13 species) or 1993 (15 species). Survival and height were recorded up to nine years after planting.

A replicated study in 1999–2000 in a degraded peat swamp in Malaysia (3) reported that 50–92% of planted tree seedlings survived over 14 months. No statistical tests were carried out. Of the six planted species, survival was highest for *Ganua motleyana* (92%) and lowest for *Calophyllum ferrugineum* (56%). Of four different planting techniques used, three supported high survival rates (82–83%, averaged across species). The fourth technique, adding oil palm fruits as mulch, supported lower survival (50%, averaged across species). In June 1999, tree seedlings were planted into 72 plots (12 plots/species) in a degraded, open peat swamp. Three plots/species were planted using each technique: mulching with oil palm fruit, planting additional nurse trees, adding topsoil or no additional intervention. In all plots, 16 seedlings were planted 3 m apart, existing vegetation was cleared and fertilizer was added to the planting holes. Survival was recorded in August 2000.

A replicated study in 1990–2002 in four historically mined bogs in eastern Canada (4) reported that 9–100% of planted tree saplings survived over 1–13 years. Five species were planted. Jack pine *Pinus banksiana* survival was 100% after two years. Tamarack *Larix laricina* survival was 52–100% over 1–9 years. Black spruce *Picea mariana* survival was 65–94% over 1–13 years. Red maple *Acer rubrum* survival was 72% after three years. Poplar *Populus* spp. survival was 9% after three years. Between 1990 and 2001, tree saplings were planted into bare peat, in single species blocks (1,600–2,500 stems/ha), in up to four separate bogs. Additional management (e.g. soil preparation, fertilization, planting density) differed between sites (see original paper). In 2002, survival of saplings (18–360/species/bog) was recorded in quadrats distributed evenly across the planted blocks.

A replicated study in 2004–2005 in a historically mined bog in Quebec, Canada (5) reported that 81% of planted tamarack *Larix laricina* seedlings and 55% of planted black spruce *Picea mariana* seedlings survived over 16 months. Additional fertilized seedlings had higher survival: 92–98% for tamarack and 58–87% for black spruce. In June 2004, seedlings were planted 3 m apart into drained, bare peat. For each species, three plots of 50 seedlings were fertilized, but three plots of 900 seedlings were not. Survival was checked after two growing seasons in October 2005.

A study in 2003–2004 in a fire-damaged peat swamp forest in Indonesia (6) reported that most planted tree seedlings survived over three months, but most had died after five months following flooding. Nine species of peat swamp trees were planted. After three months, 65–85% of seedlings had survived. However, after five months <5% of seedlings remained alive following unusually deep flooding. At this point, survival was highest for myrtle *Eugenia* sp. (27%) and red lauan *Shorea pauciflora* (13%). No *Palaquium* sp., *Gluta wallichii* or *Dryera polyphylla* seedlings survived the deep flooding. In November and December 2003, fourteen thousand tree seedlings were planted into individual mounds (30–50 cm tall) within a burned peat swamp. Most seedlings (94%) were *Gonystylus bancanus*, *Palaquium* sp., *Gluta wallichii* and *Shorea pacuiflora*. Survival was monitored in February and, for 10% of the planted mounds, in April 2004.

A study in 2002–2006 in a logged peat swamp in Kalimantan, Indonesia (7) reported that 83% of planted red balau *Shorea balangeran* seedlings survived over 40 months. On average, these were 206 cm tall and had 27 cm diameter stems. Additional seedlings inoculated with root fungi had higher growth rates than the uninoculated seedlings with three of three fungal species (213–240 cm tall; 30–37 cm diameter), but higher survival (85%) in only one of three cases. In November 2002, 100 red balau seedlings were planted (1 m apart) into logged forest. Seedlings had been grown in sterilized peat in a nursery. One hundred seedlings inoculated with each of three fungal species were also planted for comparison. Seedling height, stem diameter and survival were measured 40 months after planting.

A replicated before-and-after study in 2004–2008 three historically mined bogs in eastern Canada (8) reported that 63–100% of planted shrub clumps survived over four years, and that survivors had grown in diameter. Survival of bog cranberry *Vaccinium oxycoccos* was 100%, crowberry *Empetrum* spp. 83%, lingonberry *Vaccinium vitis-ideae* 71%, and mixed-species clumps (mostly *Vaccinium* berry species) 63%. Approximately 96% of surviving clumps showed positive growth (data not reported for single species). Amongst these, diameter increased by 50 cm/year for bog cranberry, 8 cm/year for crowberry, 22 cm/year for lingonberry and 7–8 cm/year for mixed clumps. Additionally, across all studied species, bigger clumps were more likely to survive (see original paper). In 2004, 916 clumps of shrub seedlings were planted, 1–2 m apart, along transects on wet peat. Initial clump diameter was recorded. Seedlings had been grown in a greenhouse from seeds in berries or scat fragments. In 2008, survival and final clump diameter were measured.

A replicated before-and-after study in 2007–2009 in a peat swamp in Indonesia (9) reported that 75–91% of planted red balau *Shorea balangeran* and jelutong *Dyera polyphylla* survived over one year, and that surviving trees had grown. Survival was not reported separately for the two planted species. After one year, planted seedlings of both species had increased in height (red balau by 4–11 cm; jelutong: by 2–4 cm) and diameter (both species by 0.6–2.7 mm). Amongst planted seedlings, growth (but not survival) differed between forest types. Seedlings grew significantly taller and thinner in closed forest vs open forest (see original paper). Inoculation with fungi had no significant effect on survival or growth (see Section 13.17). In 2007 or 2008, nursery-reared seedlings (800 red balau and 700 jelutong) were planted in five forest types from natural/closed forest to degraded/open land. Between half and two-thirds of the seedlings had been inoculated with root fungi. After one year, all seedlings' survival and growth were measured.

A replicated before-and-after study in 2007 in a burned peat swamp forest in Sabah, Malaysia (10) reported that two of three planted tree species survived and grew in height and diameter over 10 weeks. Of 15 planted geronggang *Cratoxylum* *arborescens* seedlings, 93% survived. For pulai *Alstonia spathulata*, survival was 87%. For katok *Stemonurus scorpioides*, survival was 0%. Geronggang seedlings increased in height by 24 cm and diameter by 2.8 mm. Pulai seedlings increased in height by 9 cm and diameter by 1.9 mm. In September 2007, three burned plots were planted with 45 seedlings (15 seedlings of each species, mixed together but 3 m apart). Survival, height and diameter of seedlings were measured over 10 weeks after planting.

A study in peat swamps in Indonesia (11) reported that planted jelutong trees (probably *Dryera polyphylla*, but not clearly reported) grew in height and diameter within a range of agroforestry systems. Over approximately six years after planting, tree height increased by 87–128 cm/year, and diameter increased by 1.6–2.2 cm/year. Trees (probably saplings, but not clearly reported) were planted in peat swamps between strips of crops, mixed with crops or amongst aquaculture operations. Tree diameter and height were measured between 64 and 78 months after planting.

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12.17 Add peatland vegetation to peatland surface Ⓑ Ⓕ S

Background

This section considers introducing peatland vegetation by spreading living vegetation onto the peatland surface, which is expected to take root and grow. It may also be possible to establish peatland vegetation on non-peat soil (e.g. mesocosm experiment of Borkenhagen & Cooper 2016). Vegetation can be sourced from nearby natural peatlands or grown in greenhouses. Direct introduction of peatland vegetation might be necessary in severely degraded or bare peatlands. Natural revegetation (from remnant plants, seed banks or dispersal) might not happen, might be very slow or might not produce the desired mix of species.

CAUTION: Collecting vegetation from natural peatlands damages the donor site, although rapid recovery has been reported (Rochefort & Campeau 2002). Non-native or non-peatland species could be present in the introduced vegetation.

This section separately considers (a) adding mosses or moss fragments to the peatland surface and (b) spreading mixed vegetation onto the peatland surface (including seeds, rhizomes, seedlings and spores of other species even if dominated by mosses). Studies that described adding individual moss plants, or that separated out mosses from mixed vegetation before sowing, are placed in the first section. Studies that collected plant material in bulk from the surface of peatlands are placed in the second section. Even if mosses are targeted or dominant, the collected material will contain seeds, rhizomes, seedlings and spores of other species (Rochefort et al. 2003).

Related interventions: directly plant whole plants into peatland (Section 12.16); introduce seeds or vegetation fragments containing seeds (Section 12.18); restoration using more than three interventions, sometimes including planting (Sections 12.1 and 12.2); supporting interventions from this section, e.g. mulching or fertilizing, used without introducing vegetation (Sections 12.3–12.14); experimental tests of interventions to complement planting (Chapter 13).

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Rochefort L., Quinty F., Campeau S., Johnson K. & Malterer T. (2003) North American approach to the restoration of *Sphagnum* dominated peatlands. *Wetlands Ecology and Management*, 11, 3–20.

12.17.1 Add mosses to peatland surface

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- **Thirteen studies** examined the effect of adding mosses or moss fragments onto peatland surfaces. Eleven were in bogs[1,2,3,4,5,8a,8b,9,10a,10b,10c] and two in were in fens[6,7]. One study[3] was a continuation of an earlier study[2]. Three of the studies[10a,10b,10c] involved sowing moss in gel beads.
- ***Sphagnum* moss cover (12 studies):** Eleven studies in bogs in the UK[1,10a,10b,10c], Canada[2,3,8a,8b], Finland[4] and Germany[9] and fens in the USA[7] reported that *Sphagnum* moss was present, after 1–4 growing seasons, in at least some plots sown with *Sphagnum*. Cover ranged from negligible to >90%. Six of the studies[3,8a,8b,10a,10b,10c] were controlled. All six found that *Sphagnum* cover or abundance was higher in sown than unsown plots. One of the studies[7] reported that *Sphagnum* only survived in one of three sites, and only when plots were mulched. One additional study in Canada[5] found that adding *Sphagnum* to bog pools did not affect *Sphagnum* cover.
- **Other moss cover (4 studies):** Four studies (including one replicated, randomized, paired, controlled, before-and-after) in bogs in Canada[8a,8b] and fens in Sweden[6] and the USA[7] reported that mosses or bryophytes other than *Sphagnum* were present, after 2–3 growing seasons, in at least some plots sown with moss fragments. Cover ranged from negligible to 76%. In the fens in Sweden[6] and the USA[7], moss cover was low (<1%) unless the plots were mulched, shaded or limed.

A before-and-after study in 1991–1993 in a historically mined raised bog in England, UK (1) reported that most *Sphagnum* moss species did not survive when sown onto peat or into pools, but that the surviving species typically spread. Of eight *Sphagnum* species spread onto bare peat, only one survived after 30 months: feathery bog moss *Sphagnum cuspidatum*. There were 20 plants/100 cm^2. Of eight *Sphagnum* species spread onto floating rafts, three survived: feathery bog moss, recurved bog moss *Sphagnum recurvum* and lobed bog moss *Sphagnum auriculatum*. There were 25–40 plants/100 cm^2. Two species had spread beyond the initial planted area. In May 1991, pairs of pools (4 m^3) and bare peat plots (4 m^2) were excavated (number of pools/plots not reported). Individual *Sphagnum* plants (5 cm long) were placed on the bare peat and on a floating mesh raft (50 plants in a 0.5 m^2 area for each species). In November 1993, survival and density of each *Sphagnum* species were recorded.

A replicated before-and-after study in 1993 in a historically mined raised bog in Quebec, Canada (2; part of 3) reported that plots sown with *Sphagnum* moss fragments developed some *Sphagnum* cover. Before sowing, plots were bare peat. After one growing season, sown plots had 1–7% *Sphagnum* cover. There were also more *Sphagnum* shoots after one growing season (180–860/m^2) than the number introduced (150–450/m^2). Additionally, cover was significantly higher in plots sown at higher densities (low initial density: 1–2%; medium: 2–4%; high: 3–7% final cover) and differed between species (see original paper). The size of introduced fragments had no effect on cover (data not reported). In June 1993, twenty 10 m^2 plots were established on bare rewetted peat. Sixteen plots were sown with a single *Sphagnum* species (four plots x four species) and four plots sown with a mixture of all four species. Within each plot, three fragment densities (low: 150; medium: 300; high: 450/m^2) and two fragment sizes (1 or 2 cm) were applied to six subplots. Additional subplots were left unsown as controls, but data were not reported. All plots were shaded with a plastic cloth. In October 1993, *Sphagnum* cover was visually estimated and live shoots counted in four 25 x 25 cm quadrats/subplot.

A replicated, randomized, paired, controlled, before-and-after study in 1993–1995 in a historically mined raised bog in Quebec, Canada (3; a continuation of 2) reported that plots sown with *Sphagnum* mosses had greater *Sphagnum* cover, over three growing seasons, than unsown plots. *Sphagnum* cover was 1–5% in sown plots but <0.5% in unsown plots. Amongst sown plots, *Sphagnum* was still present after the third, driest growing season. However, cover had dropped to 1–3%. These results were not based on tests of statistical significance. *Sphagnum* cover was significantly higher in plots sown at higher densities (low initial density: 1%; medium: 2%; high: 3% final cover) but was not affected by the size of introduced fragments (data not reported). In June 1993, twenty 10 m^2 plots were established on rewetted bare peat. In each plot, six subplots were sown with *Sphagnum* moss fragments whilst two subplots were not sown. Amongst the sown subplots, three fragment densities (low: 150; medium: 300; high: 450/m^2) and two fragment sizes (1 or 2 cm) were applied. All plots were shaded with a plastic cloth. In October 1993, *Sphagnum* cover was estimated and live shoots counted in four 25 x 25 cm quadrats/subplot.

A replicated before-and-after study in 1994–1998 in a historically mined bog in Finland (4) reported that plots sown with fragments of fine bog moss *Sphagnum angustifolium* (after rewetting) developed cover of fine bog moss. Before sowing, plots were bare peat. After four years, fine bog moss cover was 29%. In the previous three years, fine bog moss cover varied between 16 and 26%. In September 1994, 2–3 cm fragments of fine bog moss were spread on six 60 x 60 cm plots, forming a covering layer. The bare peat plots had been rewetted earlier in 1994. The moss was collected from a nearby pristine bog. Every August from 1995 to 1998, *Sphagnum* cover was measured in one 30 cm^2 quadrat/plot.

A replicated, controlled, before-and-after study in 1999–2003 in a historically mined bog in eastern Canada (5) found that sowing *Sphagnum* moss (and herbs) into created pools did not significantly affect vegetation cover. After four years, planted and unplanted pools had similar cover of *Sphagnum* (13 vs 9%), other mosses (3 vs 3%), herbs (3 vs 5%) and shrubs (6 vs 5%). In 1999, eight 6 x 8.5 m pools were created by excavating and rewetting a bog (blocking ditches and building embankments). In 2000, four pools were sown with *Sphagnum* moss (introduced to the water column). Four herb species were also planted in and around these pools. The other four pools were not planted, although bog vegetation fragments were spread onto the rest of the peatland (see Section 12.17.2). In 2003, vegetation cover was recorded in 36 quadrats/pool, each 30 x 30 cm, along six bank-to-bank transects.

A replicated before-and-after study in 2004–2005 in a degraded fen in Sweden (6) reported that four sown fen-characteristic moss species had variable survival after one growing season, and developed variable cover after two growing seasons. Before sowing, plots were bare peat. One growing season after sowing, 4–93% of moss fragments had survived. Two growing seasons after planting, cover of fen-characteristic mosses was <1–34%. Additionally, survival and cover were significantly higher in limed than unlimed plots (See Section 13.1) and in plots covered with mulch or plastic gauze than uncovered plots (see Section 13.5). In June 2004, fragments of four fen-characteristic moss species were added to 24 plots (625 cm^2) of bare rewetted peat: two scorpion mosses *Scorpidium* spp., three-ranked spear moss *Pseudocalliergon trifarium*, and starry feather moss *Campylium stellatum*. Each species was sown in separate 9 cm^2 subplots (number not reported; density 16 fragments/subplot). Twelve plots were also limed and eight were covered (with sedge litter or plastic gauze). After one growing season, moss survival was assessed in each subplot. After two growing seasons, moss cover was visually estimated.

A replicated before-and-after study in 2007–2010 in three degraded fens in Colorado, USA (7) reported that mosses established in 4 of 12 plots sown with moss fragments, and only when mulched. Before sowing, plots were bare peat. After three years, no moss survived on six plots without mulch. Under mulch, Russow's bog moss *Sphagnum russowii* survived in one of three sites (reaching 19% cover) and haircap moss *Polytrichum strictum* survived in all three sites (reaching 3–11% cover). In July 2007, moss fragments (<1 cm length) were spread onto twelve bare peat plots in each fen. Moss was a mixture of three *Sphagnum* species and haircap moss. Of the twelve plots, six were mulched with straw (immediately) and shredded aspen (after one year). In summer 2010, moss cover was measured using a pin-drop quadrat.

A replicated, randomized, paired, controlled, before-and-after study in 2007–2010 in two historically disturbed bogs in Ontario, Canada (8a) found that plots sown with *Sphagnum* moss fragments had greater bryophyte cover, after three years, than unsown plots. This was true for both *Sphagnum* moss cover (sown: 38–52%; unsown: 8%) and total bryophyte cover (sown: 66–76%; unsown: 26%). Amongst sown plots, bryophyte cover did not significantly differ between plots with and without mulch (see Section 13.4), nurse plants (see Section 13.6) or peat blocks for shelter (see Section 13.10). In August 2007, forty-eight 2 x 2 m plots were established, in six blocks of eight, across two bogs. Plots were initially bare peat, following disturbance from vehicles or pipeline construction. Forty-two plots (seven random plots/block) were sown with fresh moss fragments (mix of rusty bog moss *Sphagnum fuscum* and flat-topped bog moss *Sphagnum fallax*). The remaining six plots (one plot/block) were not sown. All plots received 30 g/m^2 rock phosphate fertilizer. Some sown plots were also mulched, sheltered with peat blocks or planted with nurse plants. In August 2010, moss cover was visually estimated in six 12.5 x 12.5 cm quadrats/plot.

A replicated, controlled, before-and-after study in 2007–2010 in two historically disturbed bogs in Ontario, Canada (8b) found that plots sown with *Sphagnum* moss fragments had greater bryophyte cover, after three years, than unsown plots. This was true for both *Sphagnum* moss cover (sown: 5–17%; unsown: 1%) and total bryophyte cover (sown: 24–51%; unsown: 21%). Adding mulch did not significantly affect bryophyte cover (see Section 13.4). In May 2007, twenty-four 1 m^2 plots of bare peat were sown with moss fragments (a mix of rusty bog moss *Sphagnum fuscum* and flat-topped bog moss *Sphagnum fallax*, stored outside during the preceding winter). Twelve of the plots were also mulched with straw. Some additional control plots (number not reported) were neither sown nor mulched. All plots received 30 g/m^2 rock phosphate fertilizer. In August 2010, moss cover was visually estimated in six 12.5 x 12.5 cm quadrats/plot.

A replicated before-and-after study in 2004–2009 and 2011–2013 in two bogs in Germany (9) reported that plots sown with *Sphagnum* moss fragments (then mulched) developed high *Sphagnum* cover. Before sowing, plots were bare peat. In one bog (Ramsloh), papillose bog moss *Sphagnum papillosum* reached 92% cover four years after spreading. In the other bog (Rastede), blunt-leaved bog moss *Sphagnum palustre* had reached 97% cover and papillose bog moss 91% cover two years after initial spreading. In 2004 (Ramsloh) and 2011 (Rastede), fragments of single moss species were spread onto 60–224 bare peat plots (15 x 15 or 25 x 25 cm). At Rastede, gaps were filled with additional fragments one year later. All plots were mulched with straw and a high water table was maintained. *Sphagnum* cover was estimated in each plot 1–3 times/year.

A replicated, paired, controlled, before-and-after study in 2010–2013 in a blanket bog in England, UK (10a) reported that *Sphagnum* moss established in 4 of 12 sown plots, mainly when bare fragments (rather than fragments in gel beads) were sown into existing vegetation (rather than onto bare peat). Before sowing, no *Sphagnum* was present. Of six grassy plots sown with *Sphagnum*, four contained the sown species after three years: three sown with bare *Sphagnum* fragments (251–450 *Sphagnum* clumps surviving; negligible cover) and one sown with *Sphagnum* in gel beads (two *Sphagnum* clumps surviving; negligible cover). Of six bare peat plots sown with *Sphagnum*, none contained the sown species after three years. Of 12 unsown control plots, nine contained no *Sphagnum* after three years but three, on grassy vegetation, contained 1–67 clumps. In May 2010, eighteen 25 m^2 plots were established: three blocks of three on restored grassy vegetation, and three blocks of three on bare peat. In each block, one plot was sown with bare *Sphagnum* fragments (<1 cm thick layer), one was sown with *Sphagnum* fragments in gel beads (400 beads/m^2) and one was not sown. However, all of these plots were mulched (with heather *Calluna vulgaris* brash). In August 2013, *Sphagnum* clumps were identified in each plot and their area was measured.

A replicated, paired, controlled study in 2009–2013 in a blanket bog in England, UK (10b) reported that *Sphagnum* moss established in 22 of 162 plots sown with moss/gel beads, but mainly when sown into existing vegetation (rather than onto bare peat). No statistical tests were carried out. After 1–3 years, *Sphagnum* clumps were present in 22 of 162 sown plots (1–288 clumps/plot or 0.06–18% of the number of beads sown). The survival rate was higher in plots with existing vegetation (natural: clumps present in 24% of sown plots; restored: clumps present in 15% of sown plots) than in bare peat plots (clumps present in 4% of sown plots). Forty adjacent unsown plots did not contain any *Sphagnum*. Between 2009 and 2012, gel beads containing *Sphagnum* fragments were sown onto a bog (4 m^2 plots; 400 beads/m^2). There were 1–3 plots for each combination of *Sphagnum* species (six options), sowing date (six options) and existing vegetation (three options: natural; restored grassy vegetation; bare peat). For each sowing date and vegetation type, some additional plots were left unsown. In August 2013, *Sphagnum* clumps were identified in each plot.

A replicated, paired, controlled study in 2010–2013 in a degraded, grassy blanket bog in England, UK (10c) reported that *Sphagnum* moss was present in 11 of 12 plots sown with moss/gel beads, but that cover was low. After three years, the 11 plots contained 4–98 discrete clumps of *Sphagnum* (0.25–6% of the number of beads sown). *Sphagnum* cover was <1% in all plots. Adjacent unsown plots did not contain any *Sphagnum*. In October 2010, fifteen 4 m^2 plots were established (in three blocks of five) on a degraded blanket bog dominated by purple moor grass *Molinia caerulea*. For each of four *Sphagnum* species, three plots (one plot/block) were sown with moss fragments encapsulated in gel beads (400 beads/m^2). The remaining three plots (one plot/block) were not sown. In all plots, grass was cut before sowing (litter left in place). In September 2013, *Sphagnum* clumps were identified in each plot and their area was measured.

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12.17.2 Add mixed vegetation to peatland surface Ⓑ Ⓕ S

- **Eighteen studies** examined the effect on peatland vegetation of spreading mixed vegetation onto the peatland surface. All 18 studies were in bogs (two[8,10] being restored as fens). One study[6a] was a continuation of an earlier study[1].

- **Characteristic plants (1 study):** One replicated, randomized, paired, controlled, before-and-after study in a degraded bog in Canada[8] found that adding fen vegetation increased the number and cover of fen-characteristic plant species.
- ***Sphagnum* moss cover (17 studies):** Seventeen replicated studies (five also randomized, paired, controlled, before-and-after) in bogs in Canada[1,2,3,4,5a,5b,6a,6b,6c,6d,6f,7a,7b,9,10], the USA[6e] and Estonia[11] reported that *Sphagnum* moss was present, after 1–6 growing seasons, in at least some plots sown with vegetation containing *Sphagnum*. Cover ranged from <1 to 73%. Six of the studies[1,5a,5b,6a,10,11] were controlled and found that *Sphagnum* cover was higher in sown than unsown plots. Five of the studies[4,6c,6d,6e,6f] reported that *Sphagnum* cover was very low (<1%) unless plots were mulched after spreading fragments.
- **Other moss cover (8 studies):** Eight replicated studies (seven before-and-after, one controlled) in bogs in Canada[4,6b,6c,6d,6f,7b], the USA[6e] and Estonia[11] reported that mosses or bryophytes[11] other than *Sphagnum* were present, after 1–6 growing seasons, in at least some plots sown with mixed peatland vegetation. Cover was <1–65%.
- **Vascular plant cover (10 studies):** Ten replicated studies in Canada[4,6b,6c,6d,6f,7a,7b,10], the USA[6e] and Estonia[11] reported that vascular plants appeared following addition of mixed vegetation fragments to bogs. Two of the studies[10,11] were controlled: one[11] found that vascular plant cover was significantly higher in sown than unsown plots, but one[10] found that sowing peatland vegetation did not affect herb cover.

A replicated, randomized, paired, controlled, before-and-after study in 1993 in a historically mined raised bog in Quebec, Canada (1; part of 6a) found that plots sown with vegetation from the surface of a donor bog contained more *Sphagnum* moss shoots than plots sown with deeper material and unsown plots. Before sowing, plots were bare peat. After one growing season, plots sown with *Sphagnum*-dominated vegetation from a bog surface contained more live *Sphagnum* shoots (190–890/m^2) than plots sown with material from 10–30 cm depth (10–100/m^2) and plots that had not been sown (30–120/m^2). Similar patterns were observed in a greenhouse experiment (see original paper). In June 1993, twelve blocks of four 1 m^2 plots were established on bare rewetted peat. Within each block, three random plots were sown with vegetation or material from a nearby natural peatland: from the surface (top 10 cm), from 10–20 cm depth or from 20–30 cm depth. The vegetation was dominated by one of three *Sphagnum* moss species. The fourth plot was not sown. All plots were shaded with a plastic cloth. In October 1993, *Sphagnum* shoots were counted in four 25 x 25 cm quadrats/plot.

A replicated before-and-after study in 1995 in a historically mined raised bog in Quebec, Canada (2) reported that plots sown with *Sphagnum*-dominated vegetation fragments (and mulched) developed *Sphagnum* moss cover. Before sowing, plots were bare peat. After one growing season, there were 146–629 *Sphagnum* shoots/m^2. Additionally, shoot density was significantly higher in plots reprofiled into depressions before sowing (with plastic sheeting: 629 shoots/m^2; without plastic sheeting: 469 shoots/m^2) than in plots that remained at surface level (146 shoots/m^2). In May 1995, three blocks of three 8 m^2 plots were established on bare rewetted peat. Plots were sown with vegetation fragments (mostly seven mixed *Sphagnum* moss species) freshly collected from the surface of nearby bogs. In each block, one plot was flat and two were reprofiled to be lower in the centre. The slopes of one reprofiled plot/block were covered with plastic sheets. All plots were mulched with straw after sowing. In October 1995, *Sphagnum* shoots were counted in 240 quadrats/plot, each 400 cm^2 and placed systematically.

A replicated before-and-after study in 1993–1994 in a historically mined bog in Quebec, Canada (3) reported that plots sown with *Sphagnum*-dominated vegetation fragments (after rewetting) developed some *Sphagnum* moss cover. Before sowing, plots were bare peat. After one year, *Sphagnum* cover was 3–6%. Between May and August 1993, vegetation fragments (mostly fine bog moss *Sphagnum angustifolium* and rusty bog moss *Sphagnum fuscum*) were scattered by hand onto three bare peat plots. The peat had been rewetted three months previously by digging water storage ditches. In September 1994, *Sphagnum* cover was estimated in 18 quadrats/plot, each 25 x 25 cm.

A replicated before-and-after study in 1995–1996 in a historically mined raised bog in Quebec, Canada (4) reported that plots sown with *Sphagnum*-dominated vegetation fragments (and mulched and/or roughened) developed some cover of mosses and vascular plants. Before sowing, plots were bare peat. After one year, *Sphagnum* cover was between 0.5 and 5%, other moss cover <1.5% and vascular plant cover <1.5%. Additionally, *Sphagnum* cover was significantly higher in plots mulched with straw (2–5%) than in unmulched plots (<0.5%) but was similar in roughened and smooth plots (<0.5–5% vs <0.5–2%). In May 1995, vegetation fragments (mostly *Sphagnum* moss) from the surface of a nearby bog were spread onto 24 rewetted bare peat plots (15 x 15 m). Twelve plots were also mulched with straw; twelve were not mulched. Eighteen plots had been roughened (by harrowing, ploughing or driving a bulldozer over); six were left smooth. In September 1996, vegetation cover was estimated in 36–72 quadrats/plot, each 25 x 25 cm.

A replicated, randomized, paired, controlled, before-and-after study in 1993–1994 in a historically mined bog in Quebec, Canada (5a) found that plots sown with *Sphagnum*-dominated vegetation fragments contained more *Sphagnum* moss shoots than unsown plots. This was true for all five focal *Sphagnum* species after one growing season (sown: 20–420; not sown: 17–90 shoots/m^2) and after two growing seasons (sown: 65–450; not sown: 25–60 shoots/m^2). These results are not based on tests of statistical significance. In spring 1993, fresh vegetation fragments (mostly *Sphagnum* moss) from the surface of a natural bog were added to slightly drained, bare peat plots (250 fragments/m^2). Twelve plots were sown with fragments dominated by each of five *Sphagnum* species (one random plot in each of 12 blocks). Blocks grouped plots by moisture and cover treatment (none, polythene sheet or shade screen). Twelve control plots were not sown. In autumn 1993 and 1994, all *Sphagnum* shoots were counted in two 30 x 30 cm quadrats/plot.

A replicated, randomized, paired, controlled, before-and-after study in 1993–1994 in a historically mined bog in Quebec, Canada (5b) reported that plots sown with *Sphagnum*-dominated vegetation fragments typically contained more *Sphagnum* moss shoots, after two growing seasons, than unsown plots. These results are not based on tests of statistical significance. Plots sown with fragments dominated by fine bog moss *Sphagnum angustifolium* or Magellan's bog moss *Sphagnum magellanicum* contained more *Sphagnum* shoots than unsown plots, whether irrigated or not (sown: 85–770; not sown: 50–80 shoots/m^2). Plots sown with fragments dominated by rusty bog moss *Sphagnum fuscum* contained more *Sphagnum* shoots than unsown plots only when irrigated (sown: 95; not sown: 80 shoots/m^2). Results after one growing season showed similar patterns. In spring 1993, fresh vegetation fragments (mostly *Sphagnum* moss) were added to slightly drained, bare peat plots (250 fragments/m^2). Six plots received fragments dominated by each of three *Sphagnum* species (one random plot in each of six blocks). Six control plots received no fragments. Three blocks were irrigated. In autumn 1993 and 1994, all *Sphagnum* shoots were counted in ten 30 x 30 cm quadrats/plot.

A replicated, randomized, paired, controlled, before-and-after study in 1993–1995 in a historically mined raised bog in Quebec, Canada (6a; a continuation of 1) found that plots sown with vegetation from the surface of a donor bog contained more *Sphagnum* moss shoots, after 1–3 growing seasons, than plots sown with deeper material and unsown plots. Before sowing, plots were bare peat. After one to three growing seasons, plots sown with *Sphagnum*-dominated fragments from the surface of a donor bog contained more live *Sphagnum* shoots (190–1,240/m^2) than plots sown with fragments from 10–30 cm depth (10–220/m^2) or plots that had not been sown with any fragments (10–150/m^2). In June 1993, twelve blocks of four 1 m^2 plots were established on rewetted bare peat. Within each block, three random plots were sown with vegetation fragments (dominated by a single *Sphagnum* moss species) collected from a nearby natural peatland: from 0–10 cm depth (surface), 10–20 cm depth or 20–30 cm depth. No vegetation fragments were added to the fourth plot. All plots were shaded with a plastic cloth. In autumn 1993, 1994 and 1995, *Sphagnum* shoots were counted in four 25 x 25 cm quadrats/plot.

A replicated before-and-after study in 1993–1999 in a historically mined bog in Quebec, Canada (6b) reported that plots sown with vegetation fragments (and rewetted and mulched) developed cover of mosses and vascular plants. Before sowing, plots were bare peat. After six growing seasons, *Sphagnum* moss cover was 34–52%, other moss cover 2–5% and vascular plant cover 7–11%. Plots sown with a high density of fragments had greater *Sphagnum* cover (52%) than those sown with lower densities (34%). In winter 1993/1994, twelve 10 x 12 m plots were sown with material from the surface of a nearby bog: primarily a mixture of four *Sphagnum* species. Material was sown at high, medium or low density (ratio of source to recipient surface 1:10, 1:20 or 1:30) and as complete or mechanically shredded fragments. All plots were rewetted and harrowed before introduction of plant material and mulched with straw afterwards. Between 1995 and 1999, autumn vegetation cover was visually estimated along transects, in 28–36 quadrats/plot.

A replicated before-and-after study in 1994–1996 in a historically mined bog in Quebec, Canada (6c) reported that plots sown with vegetation fragments (some also mulched) developed some cover of mosses and vascular plants. No statistical tests were carried out. Before sowing, plots were bare peat. After three growing seasons, total vegetation cover was 3–24%. *Sphagnum* moss cover was <1–7%, other moss cover 1–13% and vascular plant cover 1–15%. Amongst all plots, *Sphagnum* formed a larger proportion of the moss cover in those mulched with straw (1–30%) than in unmulched plots (<1%). In early 1994, mixed plant material was collected from a natural bog and spread onto 12 pairs of plots (each 3 x 15 m), situated on bare rewetted peat. Then, one random plot in each pair was covered in straw mulch. In 1994 and 1996, vegetation cover was estimated within quadrats in each plot (details not reported).

A replicated before-and-after study in 1994–1996 in a historically mined bog in Quebec, Canada (6d) reported that plots sown with vegetation fragments (some also covered) developed cover of mosses and vascular plants. Before sowing, plots were bare peat. After three years, total vegetation cover was 3–20%. *Sphagnum* moss cover was <1–3%, other moss cover 2–16% and vascular plant cover <1%. Plots mulched with straw had significantly higher cover of all plant groups (except vascular plants) than plots shaded with a plastic screen, plots covered with shrub roots or unprotected control plots. Amongst these other treatments, vegetation cover was similar. In spring 1994, the moss layer was scraped from the surface of a natural bog and spread onto twelve 9 m^2 bare peat plots. Three plots received each of the four cover treatments: straw, plastic, roots or none. In 1994 and 1996, vegetation cover was estimated in each plot (details not reported).

A replicated before-and-after study in 1997–1999 in a historically mined bog in Minnesota, USA (6e) reported that plots sown with vegetation fragments (some also mulched and/or planted with nurse plants) developed cover of mosses and vascular plants. Before sowing, plots were bare peat. After two growing seasons, total vegetation cover was 2–77%. *Sphagnum* moss cover was 0–73%, other moss cover 0–1% and vascular plant cover 1–3%. Plots mulched with straw had higher total vegetation and *Sphagnum* cover than unmulched plots, but similar cover of other mosses and vascular plants (see Section 13.4). Plots planted with nurse sedges had similar cover of all vegetation groups to plots without nurse sedges (see Section 13.6). These results were not tested for statistical significance. In 1997–1998, vegetation was scraped from the surface of natural bogs and spread onto forty-eight bare peat plots (1.5 x 1.5 m), arranged in six blocks of eight. Four random plots/block were mulched with straw (3,000 kg/ha). Four random plots/block were also planted with sedges *Carex oligosperma* before adding vegetation fragments. In October 1999, vegetation cover was visually estimated in four 25 x 25 cm quadrats/plot.

A replicated before-and-after study in 1993–1996 in a historically mined bog in Quebec, Canada (6f) reported that plots sown with vegetation fragments (some also mulched) developed some cover of mosses and vascular plants. Before sowing, plots were bare peat. After three growing seasons, sown and mulched plots had 3–11% total vegetation cover, 1–4% *Sphagnum* moss cover, 2–6% other moss cover and 1–2% vascular plant cover. Sown plots that were not mulched had <2% vegetation cover (a mixture of *Sphagnum*, other moss and vascular plants). In autumn 1993, vegetation was scraped from the surface of a natural bog and spread onto a ploughed, bare peat site. Within this site, some 10 x 10 m plots were mulched with straw immediately or in the following spring (number of plots not reported). In autumn 1996, vegetation cover was visually estimated in fourteen 25 x 25 cm quadrats/plot.

A replicated before-and-after study in 1996–1999 in a historically mined raised bog in Quebec, Canada (7a) reported that plots sown with *Sphagnum*-dominated vegetation fragments (then mulched) developed cover of *Sphagnum* and vascular plants. Before sowing, plots were bare peat. After four growing seasons, *Sphagnum* cover was 8–62% and vascular plant cover 5%. Additionally, plots reprofiled into basins before sowing had significantly greater *Sphagnum* cover (56–62%) than plots that remained at surface level (8–23%). Vascular plant cover did not differ between reprofiled and surface-level plots. In May 1996, freshly collected vegetation fragments (mostly *Sphagnum* moss) were sown onto eight 8 x 12 m plots. Four of these plots had been reprofiled (20–25 cm depth of peat pushed into ridges around the plot). Equally sized areas of each plot were sown with vegetation dominated by rusty bog moss *Sphagnum fuscum*, Magellanic bog moss *Sphagnum magellanicum* or red bog moss *Sphagnum rubellum*. All plots were mulched with straw after sowing. In autumn 1999, vegetation cover was estimated in 72 quadrats, each 25 x 25 cm, across each plot.

A replicated before-and-after study in 1996–1999 in a historically mined raised bog in Quebec, Canada (7b) reported that plots sown with vegetation fragments (some also reprofiled) developed cover of mosses and vascular plants. Before sowing, plots were bare peat. After four growing seasons, *Sphagnum* moss cover was 17–52%, other moss cover 2% and vascular plant cover 2–4%. Plots that had been reprofiled into basins before sowing had significantly greater *Sphagnum* cover (41–52%) than plots that remained at surface level (17–19%), but similar cover of other mosses and vascular plants. In May 1996, freshly collected vegetation fragments were spread by hand onto 14 plots. The fragments were mainly rusty bog moss *Sphagnum fuscum* or red bog moss *Sphagnum rubellum* but contained seeds and fragments of other plants. Four plots (15 x 15 m) were at surface level, whilst ten plots had been reprofiled into depressions (4–20 m wide) bordered by peat ridges (30–60 cm high). All plots were mulched with straw after sowing. In autumn 1998, vegetation cover was visually estimated in 12–30 quadrats, each 25 x 25 cm, across each plot.

A replicated, randomized, paired, controlled, before-and-after study in 2001–2002 in a historically mined bog in Quebec, Canada (8) found that plots spread with peat and vegetation from donor fens developed greater cover and richness of fen-characteristic plant species than plots without added material. Before intervention, no vegetation was present. After sixteen months, plots spread with material from local fens had greater cover of fen-characteristic plants (21–32%) than plots that were not spread with material (6–10%). Plots spread with fen material also contained more fen-characteristic plant species (10–15) than plots that were not spread (5–8). Patterns were similar six months after sowing (see original paper). Note that the aim of this study was to create a fen, as the post-mining peat chemistry was more like a fen than a bog. In May 2001, 54 plots (each 5 x 5 m) were established, in three equal blocks, on a historically mined bog. Surface vegetation and soil from moss- or grass-dominated fens was spread onto 36 plots (12 random plots/block) but not the other 18 plots (three plots/block). All plots had been prepared by rewetting, raking and fertilizing. In October 2001 and August 2002, cover of every plant species was estimated in ten 30 x 30 cm quadrats/plot.

A replicated, randomized, paired, before-and-after study in 1995–2001 in a historically mined bog in Quebec, Canada (9) reported that plots sown with *Sphagnum*-dominated vegetation fragments (and rewetted and mulched) developed *Sphagnum* cover. Before sowing, plots were bare peat. After four growing seasons, *Sphagnum* cover was 23–48%. Plots sown with vegetation dominated by rusty bog moss *Sphagnum fuscum* had significantly greater *Sphagnum* cover (48%) than plots sown with vegetation dominated by three other single species (red bog moss *Sphagnum rubellum* 34%; fine bog moss *Sphagnum angustifolium*: 30%; Magellan's bog moss *Sphagnum magellanicum*: 23%). Overall, there was no significant difference in *Sphagnum* cover between plots sown with single species (23–48%) or mixed species (32–40%). Each spring between 1995 and 1998, forty-five 30 m^2 plots were established (in five blocks of nine) on bare rewetted peat. Within each block, four random plots were sown with vegetation dominated by a single *Sphagnum* species and five were sown with vegetation containing a mixture of 2–4 species. All plots were then mulched with straw. *Sphagnum* cover was visually estimated each autumn, for four years after sowing.

A replicated, randomized, paired, controlled, before-and-after study in 2004–2006 in a historically mined bog in Quebec, Canada (10) found that plots sown with vegetation fragments developed greater plant species richness and *Sphagnum* moss cover than unsown plots, but similar total vegetation and herb cover. Before sowing, plots were bare peat. After two years, sown 30 m^2 plots contained more plant species than unsown plots (24 vs 22) and had greater *Sphagnum* cover (13 vs 0%). There was no significant difference between sown and unsown plots for total vegetation cover

(41 vs 36%), total herb cover (approximately 30%) or sedge *Carex* spp. cover (3 vs 2%). The study also found greater total vegetation, *Sphagnum* and sedge cover in plots receiving vegetation from moss-dominated fens than from grass-dominated fens (see original paper). Note that the aim of this study was to create a fen, as the post-mining peat chemistry was more like a fen than a bog. In May–August 2004, vegetation fragments from moss- or grass-dominated fens were spread onto 30 cleared and levelled 5 x 6 m plots. Ten similar plots received no donor material. Some sown and unsown plots were also fertilized and mulched with straw. In September 2006, cover of every plant species was estimated in 10–20 quadrats/plot.

A replicated, controlled study in 2012–2014 in a historically mined bog in Estonia (11) found that plots sown with vegetation fragments developed greater cover of bryophytes and vascular plants than an unsown plot, and had plant communities more like the donor bog. After 1–2 years, sown plots had greater cover than an unsown plot of total bryophytes (52–65% vs 5%), *Sphagnum* mosses (50–54% vs 2%) and vascular plants (17–21% vs 12%). Sheathed cottongrass *Eriophorum vaginatum* and sedge *Carex* sp. were present in at least one sown plot (cover <1%), but not in the unsown plot. After two years, the overall plant community in sown plots was 40–67% similar to the donor bog, compared to 28–45% similarity between the unsown plot and donor bog. In spring 2012, three plots were sown with plant fragments (mostly *Sphagnum* mosses) from the surface of a nearby bog. One additional plot was not sown. All plots had been reprofiled (top 20 cm of peat pushed into ridges around the plot) and rewetted and were mulched with straw. In June and September 2013 and 2014, vegetation cover was estimated in ten 50 x 50 cm quadrats/plot.

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(2) Bugnon J.-L., Rochefort L. & Price J.S. (1997) Field experiment of *Sphagnum* reintroduction on a dry abandoned peatland in eastern Canada. *Wetlands*, 17, 513–517.

(3) LaRose S., Price J. & Rochefort L. (1997) Rewetting of a cutover peatland: hydrologic assessment. *Wetlands*, 17, 416–423.

(4) Price J., Rochefort L. & Quinty F. (1998) Energy and moisture considerations on cutover peatlands: surface microtopography, mulch cover and *Sphagnum* regeneration. *Ecological Engineering*, 10, 293–312.

(5) Rochefort L. & Bastien D.F. (1998) Réintroduction de sphaignes dans une tourbière exploitée: évaluation de divers moyens de protection contre la dessiccation (Reintroduction of *Sphagnum* to an exploited bog: evaluation of various methods for protection against desiccation; in French). *Écoscience*, 5, 117–127.

(6) Rochefort L., Quinty F., Campeau S., Johnson K. & Malterer T. (2003) North American approach to the restoration of *Sphagnum* dominated peatlands. *Wetlands Ecology and Management*, 11, 3–20.

(7) Campeau S., Rochefort L. & Price J.S. (2004) On the use of shallow basins to restore cutover peatlands: plant establishment. *Restoration Ecology*, 12, 471–482.

(8) Cobbaert D., Rochefort L. & Price J.S. (2004) Experimental restoration of a fen plant community after peat mining. *Applied Vegetation Science*, 7, 209–220.

(9) Chirino C., Campeau S. & Rochefort L. (2006) *Sphagnum* establishment on bare peat: the importance of climatic variability and *Sphagnum* species richness. *Applied Vegetation Science*, 9, 285–294.

(10) Graf M.D. & Rochefort L. (2008) Techniques for restoring fen vegetation on cut-away peatlands in North America. *Applied Vegetation Science*, 11, 521–528.

(11) Karofeld E., Müür M. & Vellak K. (2016) Factors affecting re-vegetation dynamics of experimentally restored extracted peatland in Estonia. *Environmental Science and Pollution Research*, 23, 13706–13717.

12.18 Introduce seeds of peatland plants

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Background

This section considers introducing seeds of peatland vegetation: directly sowing seeds, spreading vegetation fragments on the peatland that will not themselves grow but contain seeds (e.g. hay harvested from fens), or adding peat/soil that contains seeds of peatland plants. Adding vegetation fragments may have the additional benefit of protecting the peat surface, controlling moisture levels and providing shade (see Section 12.11). Note that this section does not include studies that only introduce, or report responses of, nurse plants (e.g. Caporn *et al.* 2007).

Direct introduction of peatland vegetation might be necessary in severely degraded or bare peatlands. Natural revegetation (from remnant plants, seed banks or dispersal) might not happen, might be very slow or might not produce the desired mix of species.

CAUTION: Collecting seeds from natural peatlands may damage the source site. Trees and shrubs are not natural features of all peatlands: growth of woody plants is often a threat to bogs and fens. Trees could dry out peatlands by taking up water.

Related interventions: directly plant whole plants into peatland (Section 12.16); spread vegetation onto peatland surface (Section 12.17); restoration using more than three interventions, sometimes including planting (Sections 12.1 and 12.2); supporting interventions from this section, e.g. mulching or fertilizing, used without introducing vegetation (Sections 12.3–12.14); experimental tests of interventions to complement planting (Chapter 13).

Caporn S., Sen R., Field C., Jones E., Carroll J. & Dise N. (2007) *Consequences of Lime and Fertilizer Application for Moorland Restoration and Carbon Balance*. Moors for the Future Research Report.

12.18.1 Introduce seeds of peatland herbs

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- **Ten studies** examined the effect, on peatland vegetation, of introducing seeds of herbaceous peatland plants. Seven studies were in fens or fen meadows[2,3,5,6,7,9,10], two in bogs[1,4] and one in unspecified peatland[8].
- **Germination (2 studies):** Two replicated studies (one also controlled, before-and-after) reported that some planted herb seeds germinated. In a bog in Germany[1] three of four species germinated, but in a fen in the USA[2] only one of seven species germinated.
- **Characteristic plants (3 studies):** Three studies (two controlled) in fen meadows in Germany[3,6] and a peatland in China[8] reported that wetland-characteristic[3,8] or peatland-characteristic[3,6] plants colonized plots where herb seeds were sown (sometimes[8] along with other interventions).
- **Herb cover (4 studies):** Three before-and-after studies (one also replicated, randomized, paired, controlled) in a bog in New Zealand[4], fen meadows in Switzerland[5] and a peatland in China[8] reported that plots sown with herb seeds developed cover of the sown herbs (and in New Zealand[4], greater cover than unsown plots). In China[8], the effect of sowing was not separated from the effects of other interventions. One replicated, randomized, paired, controlled study in a fen in the USA[9] found that plots sown with herb (and shrub) seeds developed similar herb cover to plots that were not sown.
- **Overall vegetation cover (3 studies):** Of three replicated, controlled studies, one in a fen in the USA[9] found that sowing herb (and shrub) seeds increased total vegetation cover. One study in a bog in New Zealand[4] found that sowing herb seeds had no effect on total vegetation cover. One

study in a fen meadow in Poland[7] found that the effect of adding seed-rich hay depended on other treatments applied to plots.

- **Overall plant richness/diversity (4 studies):** Two replicated, controlled studies in fens in the USA[9] and Poland[10] found that sowing herb seeds had no effect on plant species richness (total[9] or vascular[10]). Two replicated, controlled, before-and-after studies in a bog in New Zealand[4] and a fen meadow in Poland[7] each reported inconsistent effects of herb sowing on total plant species richness.

A replicated, controlled, before-and-after study in 1993–1995 in a historically mined raised bog in Germany (1) reported that planted seeds germinated for three of four herb species, and that seedlings survived over two years. After two years, seedling densities (seedlings/400 cm^2) were 8–45 for purple moor grass *Molinia caerulea*, 4–10 for sheathed cottongrass *Eriophorum vaginatum*, 0–1 for common cottongrass *Eriophorum angustifolium* and 0 for beaked sedge *Carex rostrata* (i.e. it did not germinate). In unsown plots, there were 0–3 seedlings/400 cm^2. Plots covered with mulch, fleece or fibre mats after sowing contained more moor grass and sheathed cottongrass seedlings (14–45 seedlings/400 cm^2) than uncovered plots (0–8 seedlings/400 cm^2). In autumn 1993, seeds of the four species were spread onto 1 m^2 plots of bare rewetted peat (20 plots/species; 40–48 seeds/400 cm^2). Five plots/species were covered with mulch, five with synthetic fleece and five with jute fibre mat, whilst five were not covered. Fifteen additional plots were not seeded (but some were covered). Seedlings were counted in summer 1994 (two plots/treatment) and 1995 (three plots/treatment).

A replicated study in 1992 in a historically mined fen in Colorado, USA (2) reported that planted seeds germinated for one of seven herb species. Arrowgrass *Triglochin maritima* germinated in 15 of 25 plots (but not the very wettest or very driest). No seeds germinated for three *Carex* sedge species, common cottongrass *Eriophorum angustifolium*, elk sedge *Kobresia simpliciuscula* or Rocky Mountain iris *Iris missouriensis*. In June 1992, each species was planted into 25 separate 0.5 x 0.5 m plots (20 seeds/plot). Plots contained shallow surface peat (a "few" centimetres), had variable water levels and had been cleared of existing vegetation. Seeds were collected from the wild in 1991 and kept cold over winter. Seeds were watered after planting. Germination and seedling survival were recorded weekly until the end of August 1992.

A controlled, before-and-after study in 1991–1997 in a degraded fen meadow in Germany (3) reported that adding seed-rich hay, after removing topsoil, ensured that plots developed wetland-characteristic plant communities. Over six years, plots with hay added after removal of 20–40 cm of topsoil developed cover of fen-characteristic herbs, including sedge *Carex* spp. and purple moor grass *Molinia caerulea*. Plots with hay added after removal of 60 cm of topsoil developed cover of wetland-characteristic herbs (particularly rushes) in addition to fen-characteristic species. Plant communities in plots without added hay showed similar changes to those with hay when 40–60 cm of topsoil was removed, but developed cover of species from drier grasslands when 20 cm of topsoil was removed. All data were reported as a graphical analysis. The results were not tested for statistical significance. In February 1991, six 4,500 m^2 plots in a historically farmed fen meadow were stripped of topsoil (to 20, 40 or 60 cm depth). Hay was cut from nearby fens and spread onto three of the plots (one stripped to each depth). From 1992 to 1997, vegetation cover was recorded annually in five 4 m^2 quadrats/plot.

A replicated, randomized, paired, controlled, before-and-after study in 1998–2000 in a historically mined raised bog in New Zealand (4) reported that plots seeded with bamboo rush *Sporadanthus ferrugineus* typically developed greater rush cover than unseeded plots, but that seeding had no consistent effect on total vegetation cover or species richness. Most of these results are not based on tests of statistical significance. After 810 days, seeded plots had greater bamboo rush cover in 8 of 12 comparisons (for which seeded: 2–32%; unseeded: 0–8% cover) and lower bamboo rush cover in only three comparisons (seeded: 0–3%; unseeded: 4–27%). Total vegetation cover increased similarly in seeded and unseeded plots, with no significant difference after 810 days (seeded: 38%; unseeded: 40%). Seeded plots contained fewer plant species than unseeded plots in 5 of 12 comparisons but more in 4 of 12, with no difference in the other three. In March 1998, forty-eight plots (25 m^2) were established, in six blocks of eight, on bare rewetted peat. Hundreds of bamboo rush seeds (1 g) were spread on twenty-four plots (four random plots/block). Twenty-four control plots were not sown. Some plots were fertilized. Canopy cover of every plant species was estimated every 1–3 months until June 2000.

A replicated before-and-after study in 2004–2005 in two degraded fen meadows in Switzerland (5) reported that plots sown with fen plant seeds (some also ploughed or mulched) developed cover of fen-characteristic plants. Before sowing, plots were bare peat. After 2–10 months, cover of the sown fen plants was 10–45%. There were 50–160 individual plants/m^2 (except in one site, where plots mulched with straw after sowing contained only 2 plants/m^2). Vegetation cover and plant density did not significantly differ between plots sown in autumn and spring (see original paper). Forty-eight 2 x 2 m plots were established across two fen meadows (historically cultivated, but stripped of topsoil before the study began). Seeds were sown (10 different species; 200–800 seeds/species/plot) in October 2004 (8 plots), April 2005 (16 plots) or June 2005 (24 plots). Some random plots were ploughed before sowing or mulched afterwards. In August 2005, individual plants were counted and total vegetation cover estimated in the central 1.5 x 1.5 m of each plot.

A replicated, paired, controlled study in 2001–2005 in a degraded fen meadow in Germany (6) found that in plots spread with hay from nearby fens, peatland-characteristic plants were more abundant than in plots without added hay. Over five years following hay addition, peatland-characteristic plants occurred in up to 28% of quadrats with up to 12% cover in each plot. In plots without added hay, peatland-characteristic plants occurred in <5% of quadrats with negligible cover. Amongst plots with added hay, abundance and cover were higher in those that had their topsoil removed prior to hay addition, but grazing had no additional effect (reported as a statistical model result). In 2001, thirty-two 6 x 6 m plots (in four blocks of eight) were established in a drained, abandoned, nutrient-enriched fen meadow. Freshly cut, seed-rich hay from an adjacent fen was added to 16 of the 32 plots. Additionally, four plots with hay and four plots without received each of the following treatments: topsoil stripping (30 cm depth) before hay addition, grazing (open to cattle) after hay addition, topsoil stripping plus grazing, or neither topsoil stripping nor grazing. Annually between 2002 and 2005, cover of every plant species was estimated in each plot, in 16 permanent 1 m^2 quadrats.

A replicated, controlled, before-and-after, site comparison study in 2004–2007 in a degraded fen meadow in Poland (7) found that the effect of adding hay on the meadow vegetation depended on other treatments applied to the plots: topsoil stripping and fencing. Overall, plots with and without added hay developed different plant communities over three years. However, only plots where hay was added after deep soil stripping developed a plant community similar to a target fen meadow (data reported as a graphical analysis; similarity not tested for statistical significance). The effect of hay addition on plant species richness, vascular plant cover, moss cover and vegetation biomass also depended on the other treatments (reported as statistical model results). For example, hay addition increased plant species richness in fenced plots (hay added: 23 species; no hay added: 18 species/4 m^2 after three years) but had no effect in unfenced plots (data not reported). In 2004 and 2005, fen meadow hay was spread (5–7 cm thick) onto 4 m^2 plots (number not clear). Some additional plots did not receive hay. All plots were historically drained, but were stripped of topsoil (20 cm or 40 cm depth) before adding hay. Half of the plots were fenced to exclude boar and deer. Vegetation cover and plant species were recorded annually between 2004 (before adding hay but after stripping soil) and 2007. Total vegetation biomass was measured from clippings taken in August 2006–2007.

A before-and-after study in historically mined peatland in China (8) reported that an area sown with seeds (also rewetted and fenced to exclude livestock) developed cover of grasses and wetland-characteristic herbs. No statistical tests were carried out. Before restoration, the peatland was largely bare peat with some sedges *Carex* spp. and herbs characteristic of drier soils (precise cover not reported). After restoration, new wetland-characteristic species had colonized, including rush *Blysmus sinocompressus* (30% cover) and marsh marigold *Caltha scaposa* (2% cover). Also abundant were tussock grass *Deschampsia cespitosa* (20% cover) and couch grass *Elymus nutans* (10% cover). Forty-two hectares of Hongyuan peatland were sown with seeds of five plant species (50 kg; mainly couch grass, other species not reported). The peatland was historically mined, drained and grazed, but had been rewetted by damming drainage ditches and fenced to exclude yaks. The study does not distinguish between the effects of these interventions. Vegetation cover was visually estimated (precise methods and dates not reported).

A replicated, randomized, paired, controlled study in 2009–2011 in a fen in Michigan, USA (9) found that plots sown with herb (and shrub) seeds developed more vegetation cover overall than unsown plots, but similar cover of all monitored plant groups and similar species richness. After two years, sown plots had greater total vegetation cover (214%) than unsown plots (165%). Cover of individual plant groups did not differ significantly between sown and unsown plots (although it tended to be higher in the former): sedges (120 vs 81%), grasses (27 vs 15%), forbs (28 vs 33%), mosses (33 vs 29%), shrubs (4 vs 4%). The same was true for plant species richness (45 vs 43 species across all quadrats). Patterns were similar, but cover lower, one year after intervention. In 2009, twenty pairs of 9 m^2 plots were established, in a ditch recently refilled with seed-rich fen spoil. Twenty plots (one random plot/pair) were sown with a mixture of herb (grasses, rushes, sedges, forbs) and shrub seeds. The other 20 plots were not sown. The study does not distinguish between the effects of sowing herbs and shrubs. In 2010 and 2011, vegetation cover was recorded in one 1 m^2 quadrat/plot.

A replicated, controlled, before-and-after study in 2008–2011 in a degraded fen in Poland (10) found that adding seed-rich hay to plots did not affect vascular plant community composition or species richness. Ten drained plots were initially dominated by dryland plants. The overall composition of the plant community did not change over two years, whether hay was added or not. Two other, wetter plots were initially dominated by fen-characteristic herbs. In these plots, rushes and reeds became more abundant over two years, whether hay was added or not (all community data reported as a graphical analysis; results not tested for statistical significance). After two years, the number of vascular plant species was not significantly different in plots with or without added hay (data not reported). In autumn 2009, twelve plots were established in a drained, degraded fen. Ten plots remained fully drained. Hay from a nearby natural fen was spread onto five of these plots. Two plots were wetter, having been been stripped of 60 cm of topsoil 6–8 months before the initial vegetation sampling. Hay was spread onto half of each of these plots. In summer 2009 (before hay addition), 2010 and 2011, cover of every vascular plant species was estimated in each plot (details not clear).

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(10) Hedberg P., Kozub Ł. & Kotowski W. (2014) Functional diversity analysis helps to identify filters affecting community assembly after fen restoration by top-soil removal and hay transfer. *Journal for Nature Conservation*, 22, 50–58.

12.18.2 Introduce seeds of peatland trees/shrubs

Ⓑ Ⓕ Ⓢ

- **Five studies** examined the effect, on peatland vegetation, of introducing seeds of peatland trees/shrubs to restore or create forested/shrubby peatland. Three studies were in bogs[1,3,4] and two were in fens[2,5].
- **Germination (2 studies):** Two replicated studies in a bog in Germany[1] and a fen in the USA[2] reported germination of heather and willow seeds, respectively, in at least some sown plots.
- **Survival (2 studies):** One replicated study in a bog in Germany[1] reported survival of some heather seedlings over two years. One replicated study in a fen in the USA[2] reported that all germinated willow seedlings died within one month.
- **Shrub cover (3 studies):** Two studies (one replicated, randomized, paired, controlled) in bogs in New Zealand[3] and Estonia[4] reported that plots sown with shrub seeds (sometimes[4] along with other interventions) developed greater cover of some shrubs than plots that were not sown: sown manuka[3] or naturally colonizing heather[4] (but not sown cranberry[4]). One replicated, randomized,

paired, controlled study in a fen in the USA[5] found that plots sown with shrub (and herb) seeds developed similar overall shrub cover to unsown plots within two years.

- **Overall vegetation cover (3 studies):** Two replicated, randomized, paired, controlled studies in a bog in New Zealand[3] and a fen in the USA[5] reported that plots sown with shrub (and herb) seeds developed greater total vegetation cover than unsown plots after two years. One site comparison study in bogs in Estonia[4] reported that sowing shrub seeds (along with fertilization) had no effect on total vegetation cover after 25 years.
- **Overall plant richness/diversity (3 studies):** One site comparison study in bogs in Estonia[4] reported that sowing shrub seeds (along with fertilization) increased plant species richness. However, one replicated, randomized, paired, controlled study in a bog in New Zealand[3] reported that plots sown with shrub seeds typically contained fewer plant species than plots that were not sown. One replicated, randomized, paired, controlled study in a fen in the USA[5] found that sowing shrub (and herb) seeds had no effect on plant species richness.

A replicated, controlled, before-and-after study in 1993–1995 in a historically mined raised bog in Germany (1) reported that planted heather *Calluna vulgaris* seeds germinated, and that some seedlings survived over two years – although survival depended on whether plots were covered. In covered plots, there were 6–24 seedlings/400 cm^2 after one year and 25–35 seedlings/400 cm^2 after two years. There were significantly fewer seedlings in uncovered plots: 5 seedlings/400 cm^2 after one year and 0–1 seedlings/400 cm^2 after two years. Unseeded plots contained no heather seedlings, whether covered or not. In autumn 1993, twenty 1 m^2 plots of bare rewetted peat were sown with heather seeds (40–48 seeds/400 cm^2). Five plots were then covered with mulch, five with synthetic fleece and five with jute fibre mat for one or two winters, whilst five were not covered. Fifteen additional plots were not sown (but some were covered). Seedlings were counted in summer 1994 (two plots/treatment) and 1995 (three plots/treatment).

A replicated study in 1992 in a historically mined fen in Colorado, USA (2) reported that planted hoary willow *Salix candida* seeds germinated in 6 of 25 plots, but that all seedlings died within one month. In June 1992, twenty fresh ripe seeds were planted into each of 25 plots (0.5 x 0.5 m, with a "few" centimetres of surface peat and variable water levels). All plots had been cleared of existing vegetation. Seeds were watered after planting. Seedling germination and survival were recorded weekly until the end of August 1992.

A replicated, randomized, paired, controlled, before-and-after study in 1998–2000 in a historically mined raised bog in New Zealand (3) reported that plots seeded with manuka *Leptospermum scoparium* typically contained fewer plant species than unseeded plots after 810 days, but had greater vegetation cover. Most of these are not based on tests of statistical significance. Seeded plots contained fewer plant species in 9 of 12 comparisons (seeded: 1–6 species; unseeded: 3–8 species). In the other three comparisons, seeded plots contained more species. Accordingly, seeded plots were dominated by dense stands of manuka, with more manuka cover than unseeded plots in 11 of 12 comparisons (seeded: 5–100%; unseeded 0–92%). Total vegetation cover increased in both seeded and unseeded plots, but was significantly higher after 810 days in the former (seeded: 52%; unseeded: 40%). In March 1998, forty-eight plots (25 m^2) were established, in six blocks, on bare rewetted peat. Manuka branches were placed on twenty-four plots (four random plots/block). The branches released seeds as they dried. Twenty-four control plots received no seeds/branches. Some plots were fertilized. Canopy cover of every plant species was estimated every 1–3 months until June 2000.

A site comparison study in two historically mined raised bogs in Estonia (4) reported that a bog sown with cranberry *Oxycoccus palustris* seeds (and fertilized) contained a different plant community to an unsown (and unfertilized) bog, with more plant species, more moss/lichen cover and more heather cover but similar total vegetation and cranberry cover. Most of these results are not based on tests of statistical significance. After 25 years, the overall plant community composition significantly differed between the sown and unsown bogs (data not reported). The sown bog contained more plant species in two of three comparisons (sown: 13–16; unsown: 7–15), greater moss/lichen cover in three of three comparisons (sown: 14–42%; unsown: 3–11%), greater *Sphagnum* moss cover in eight of nine comparisons (sown: <1–8%; unsown: 0–2%) and greater cover of heather *Calluna vulgaris* in three of three comparisons (sown: 1–28%; unsown: <1–17%). However, the bogs had similar total vegetation cover in two of three comparisons (sown: 42–46%; unsown: 41–48%) and cranberry cover in three of three comparisons (sown: <1–3%; unsown: 0–1%). In the late 1980s, one historically mined bog was sown with cranberry seeds (20 kg/ha) and fertilized (phosphate; 350 kg/ha). The study does not distinguish between the effects of these interventions. Another historically mined bog was neither sown nor fertilized. In the early 2000s, vegetation cover was assessed in 235 quadrats (1 m^2) across the bogs: placed in ditches, along ditch margins and on flat peat.

A replicated, randomized, paired, controlled study in 2009–2011 in a fen in Michigan, USA (5) found that plots sown with shrub (and herb) seeds developed more vegetation cover overall than unsown plots, but similar cover of all monitored plant groups and similar species richness. After two years, sown plots had greater total vegetation cover (214%) than unsown plots (165%). Cover of individual plant groups did not differ significantly between sown and unsown plots (although it tended to be higher in the former): shrubs (4 vs 4%), sedges (120 vs 81%), grasses (27 vs 15%), forbs (28 vs 33%), bryophytes (33 vs 29%). The same was true for plant species richness (45 vs 43 species across all quadrats). Patterns were similar, but cover lower, one year after intervention. In 2009, twenty pairs of 9 m^2 plots were established, in a ditch recently refilled with seed-rich fen spoil. Twenty plots (one random plot/pair) were sown with a mixture of shrub and herb seeds. The other 20 plots were not sown. The study does not distinguish between the effects of sowing shrubs and herbs. In 2010 and 2011, vegetation cover was recorded in one 1 m^2 quadrat/plot.

(1) Sliva J. & Pfadenhauer J. (1999) Restoration of cut-over raised bogs in southern Germany – a comparison of methods. *Applied Vegetation Science*, 2, 137–148.

(2) Cooper D.J. & MacDonald L.H. (2000) Restoring the vegetation of mined peatlands in the southern Rocky Mountains of Colorado, USA. *Restoration Ecology*, 8, 103–111.

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13. Actions to complement planting

Background

This chapter highlights interventions that can be used to complement deliberate introduction of desirable peatland plants. We include all studies that compare plots or areas where vegetation has been introduced with and without a helping intervention. Key metrics are germination rate, survival rate, growth rate and vegetation cover.

In most cases, it is the planted vegetation that responds to the helping intervention. In some cases, the intervention might affect non-planted vegetation as well or instead (e.g. fertilization might stimulate growth of seeds already in the soil, not just planted seeds). We cross-reference studies that also provide data from unplanted plots treated with the helping intervention, described in Chapter 12. These data may indicate which vegetation (planted or non-planted) was responding to the intervention.

This chapter includes studies performed in greenhouses, laboratories or nurseries if they test an intervention as it would be used in the field (e.g. adding fertilizer). Studies that change environmental conditions in other ways (e.g. altering water level by placing plant pots at different depths) are not summarized as evidence.

Related interventions: interventions in this chapter may be used without introducing vegetation e.g. adding fertilizer but not sowing seeds (Chapter 12, also Chapter 8 and Chapter 10). Chapter 12 sometimes mentions the effects of interventions from Chapter 13 as implementation options within the overall effect of planting, but the effects are described here in more detail.

Key messages

13.1 *Add lime (before/after planting)* **6 studies**

Survival: One replicated, controlled study in the Netherlands reported that liming reduced survival of planted fen herbs after two growing seasons. One replicated, randomized, paired, controlled study in Sweden found that liming increased survival of planted fen mosses over one season.

Growth: Two controlled, before-and-after studies found that liming did not increase growth of planted peatland vegetation: for two *Sphagnum* moss species in bog pools in the UK, and for most species of peat swamp tree in a nursery in Indonesia. One replicated, controlled, before-and-after study in Sweden found that liming increased growth of planted fen mosses.

Cover: Of two replicated, randomized, paired, controlled studies, one in a fen in Sweden found that liming increased cover of sown mosses. The other, in a bog in Canada, found that liming plots sown with mixed vegetation did not affect vegetation cover (total, vascular plants or bryophytes).

13.2 *Add inorganic fertilizer (before/after planting)* **9 studies**

Survival: Two replicated, randomized, paired, controlled studies in bogs in Canada examined the effect, on plant survival, of adding inorganic fertilizer to areas planted with peatland plants. One study reported that fertilizer increased survival of two planted tree species. The other study found that fertilizer had no effect on three planted tree species and reduced survival of one.

Growth: Five studies (three replicated, randomized, paired, controlled) in bogs in the UK, Germany and Canada found that fertilizer typically increased growth of planted mosses, herbs or trees.

However, for some species or in some conditions, fertilizer had no effect on growth. One replicated, randomized, controlled, before-and-after study in a nursery in Indonesia found that fertilizer typically had no effect on growth of peat swamp tree seedlings.

Cover: Three replicated, randomized, paired, controlled studies in bogs examined the effect, on vegetation cover, of adding inorganic fertilizer to areas planted with peatland plants. One study in Canada found that fertilizer increased total vegetation, vascular plant and bryophyte cover. Another study in Canada found that fertilizer increased sedge cover but had no effect on other vegetation. One study in New Zealand reported that fertilizer typically increased cover of a sown shrub and rush, but this depended on the chemical in the fertilizer and preparation of the peat.

13.3 *Add organic fertilizer (before/after planting)* — 0 studies

We captured no evidence for the effect, on peatland vegetation, of adding organic fertilizer to areas planted with peatland plants.

13.4 *Cover peatland with organic mulch (after planting)* — 12 studies

Germination: One replicated, controlled, before-and-after study in a bog in Germany found that mulching after sowing seeds increased germination of two species (a grass and a shrub), but had no effect on three other herb species.

Survival: Two replicated, paired, controlled studies in a fen in Sweden and a bog in the USA reported that mulching increased survival of planted vegetation (mosses or sedges). One replicated, paired, controlled study in Indonesia reported that mulching with oil palm fruits reduced survival of planted peat swamp tree seedlings.

Growth: One replicated, randomized, paired, controlled, before-and-after study in a fen in the USA reported that mulching increased growth of transplanted sedges.

Cover: Six studies (including four replicated, randomized, paired, controlled, before-and-after) in bogs in Canada and the USA, and a fen in Sweden, found that mulching after planting increased vegetation cover (specifically total vegetation, total mosses/bryophytes, *Sphagnum* mosses or vascular plants after 1–3 growing seasons). Three replicated, randomized, paired, controlled, before-and-after studies in bogs in Canada found that mulching after planting had no effect on vegetation cover (*Sphagnum* mosses or fen-characteristic plants).

13.5 *Cover peatland with something other than mulch (after planting)* — 8 studies

Germination: One replicated, controlled, before-and-after study in a bog in Germany reported mixed effects of fleece and fibre mats on germination of sown herb and shrub seeds (positive or no effect, depending on species).

Survival: Two replicated, randomized, controlled studies examined the effect, on plant survival, of covering planted areas. One study in a fen in Sweden reported that shading increased survival of planted mosses. One study in a nursery in Indonesia reported that shading typically had no effect on peat swamp tree survival, but increased survival of some species.

Growth: Three replicated, randomized, controlled, before-and-after studies examined the effect, on plant growth, of covering planted areas. One study in a greenhouse in Switzerland found that covers, either transparent plastic or shading mesh, increased growth of planted *Sphagnum* moss. One study in a fen in Sweden found that shading with plastic mesh reduced growth of planted fen mosses. One study in a nursery in Indonesia reported that seedlings shaded with plastic mesh grew taller and thinner than unshaded seedlings.

Cover: Two replicated and paired studies, in a fen in Sweden and a bog in Australia, reported that shading plots with plastic mesh increased planted moss cover. One study in a bog in Canada found that covering sown plots with plastic mesh, but not transparent plastic sheets, increased the number of *Sphagnum* moss shoots. Another study in a bog in Canada reported that shading sown plots with plastic mesh did not affect cover of vegetation overall, vascular plants or mosses.

13.6 *Introduce nurse plants (to aid focal peatland plants)* **3 studies**

Survival: One replicated, paired, controlled study in Malaysia reported that planting nurse trees had no effect on survival of planted peat swamp tree seedlings (averaged across six species).

Cover: Two replicated, randomized, paired, controlled, before-and-after studies in bogs in the USA and Canada found that planting nurse herbs had no effect on cover, after 2–3 years, of other planted vegetation (mosses/bryophytes, vascular plants or total cover).

13.7 *Rewet peatland (before/after planting)* **0 studies**

We captured no evidence for the effect on peatland vegetation of rewetting (by raising the water table) areas planted with peatland plants.

13.8 *Irrigate peatland (before/after planting)* **1 study**

Cover: One replicated, paired, controlled, before-and-after study in a bog in Canada found that irrigation increased the number of *Sphagnum* moss shoots present 1–2 growing seasons after sowing *Sphagnum* fragments.

13.9 *Reprofile/relandscape peatland (before planting)* **4 studies**

Survival: One replicated, paired, controlled study in a bog in Canada found that over one growing season, survival of sown *Sphagnum* mosses was higher in reprofiled basins than on raised plots.

Cover: Two replicated, controlled, before-and-after studies in bogs in Canada found that reprofiled basins had higher *Sphagnum* cover than raised plots, 3–4 growing seasons after sowing *Sphagnum*-dominated vegetation fragments. One controlled study in a bog in Estonia reported that reprofiled and raised plots had similar *Sphagnum* cover, 1–2 years after sowing. All three studies found that reprofiled and raised plots developed similar cover of other mosses/bryophytes and vascular plants.

13.10 *Create mounds or hollows (before planting)* **3 studies**

Growth: One controlled study, in a peat swamp in Thailand, reported that trees planted into mounds of peat grew thicker stems than trees planted at ground level.

Cover: Two replicated, randomized, paired, controlled, before-and-after studies in bogs in Canada found that roughening the peat surface (e.g. by harrowing or adding peat blocks) did not significantly affect cover of planted *Sphagnum* moss, after 1–3 growing seasons.

13.11 *Remove upper layer of peat/soil (before planting)* **0 studies**

We captured no evidence for the effect, on peatland vegetation, of removing the upper layer of peat or soil before planting peatland plants.

13.12 *Bury upper layer of peat/soil (before planting)* **0 studies**

We captured no evidence for the effect, on peatland vegetation, of burying the upper layer of peat or soil before planting peatland plants.

13.13 *Add fresh peat to peatland (before planting)* **1 study**

Cover: One replicated, controlled, before-and-after study in a bog New Zealand reported that plots amended with fine peat supported higher cover of two sown plant species than the original (tilled) bog surface.

13.14 *Encapsulate planted moss fragments in beads/gel* **0 studies**

We captured no evidence for the effect of encapsulating moss fragments on their performance, relative to loose moss fragments, when introduced to peatlands.

13.15 *Use fences or barriers to protect planted vegetation*	0 studies

We captured no evidence for the effect of using fences or barriers to protect planted peatland vegetation.

13.16 *Remove vegetation that could compete with planted peatland vegetation*	1 study

Survival: One controlled study in a bog in the UK reported that some *Sphagnum* moss survived when sown, in gel beads, into a plot where purple moor grass had previously been cut. No moss survived in a plot where grass had not been cut.

13.17 *Add root-associated fungi to plants (before planting)*	3 studies

Survival: Two controlled studies (one also replicated, paired, before-and-after) in peat swamps in Indonesia found that adding root fungi did not affect survival of planted red balau or jelutong in all or most cases. However, one fungal treatment increased red balau survival.

Growth: Two replicated, controlled, before-and-after studies of peat swamp trees in Indonesia found that adding root fungi to seedlings, before planting, typically had no effect on their growth. However, one controlled study in Indonesia found that adding root fungi increased growth of red balau seedlings.

13.18 *Protect or prepare vegetation before planting (other interventions)*	0 studies

We captured no evidence for the effect of protecting or preparing peatland vegetation before planting (other than by adding root-associated fungi).

Interventions

13.1 Add lime (before/after planting) Ⓑ Ⓕ Ⓢ

- **Six studies** examined the effect, on peatland vegetation, of liming areas planted with peatland plants. Four studies involved fen plants[2,3a,3b,5], one involved bog plants[1] and one involved tropical peat swamp plants[4]. Two of the studies were in greenhouses/nurseries[3b,4].
- **Survival (2 studies):** One replicated, controlled study in the Netherlands[2] reported that liming typically reduced survival of planted fen herbs after two growing seasons. One replicated, randomized, paired, controlled study in Sweden[3a] found that liming increased survival of planted fen mosses over one growing season.
- **Growth (3 studies):** Two controlled, before-and-after studies found that liming typically did not increase growth of planted peatland vegetation. Liming reduced or had no effect on *Sphagnum* moss growth in bog pools in the UK[1], and reduced growth rates for the majority of peat swamp tree seedlings in a nursery in Indonesia[4]. One replicated, controlled, before-and-after study in Sweden[3b] found that liming increased growth of planted fen mosses.
- **Cover (2 studies):** One replicated, randomized, paired, controlled, before-and-after study in a fen in Sweden[3a] found that liming increased cover of sown mosses. However, one replicated, randomized, paired, controlled study in a bog in Canada[5] found that liming plots sown with fen vegetation fragments had no effect on total vegetation, vascular plant or bryophyte cover.

Background

Peatland plant survival and growth is partly determined by the acidity of a peatland, or pH (Rydin & Jeglum 2013). Fen plants grow in alkaline to weakly acidic peat (approximately pH 6–8, similar to saliva, tap water or sea water). Bog plants grow in more acidic peat (approximately pH 4–5, similar to tomato juice or coffee). The acidity of a peatland also determines the availability of nutrients. For example, phosphorous becomes locked away in acidic soils (Weil & Brady 2016).

Lime (calcium and/or magnesium-rich chemicals) can be added to peatlands to reduce acidity and modify nutrient availability, potentially increasing survival or growth of peatland vegetation. To be included as evidence in this section, studies must have reported the response of peatland vegetation, not just nurse plants (e.g. Caporn *et al.* 2007).

CAUTION: The benefits and harms of liming are very context specific. Liming is mostly used in fens and fen meadows, but is sometimes used in extremely polluted, exceptionally acidic bogs. Lime could damage most bogs by reducing natural acidity.

Related interventions: add lime to peatlands without introducing vegetation (Section 10.13); restoration using multiple interventions, sometimes including liming (Section 12.1); introduce peatland vegetation – overall effects (Sections 12.16–12.18).

Caporn S., Sen R., Field C., Jones E., Carroll J. & Dise N. (2007) *Consequences of Lime and Fertilizer Application for Moorland Restoration and Carbon Balance*. Moors for the Future Research Report.

Rydin H. & Jeglum J.K. (2013) *The Biology of Peatlands, Second Edition*. Oxford University Press, Oxford.

Weil R.R. & Brady N.C. (2016) *The Nature and Properties of Soils, Fifteenth Edition*. Pearson, USA.

A controlled, before-and-after study in 1991 in a historically mined raised bog in England, UK (1) found that liming reduced growth of one planted *Sphagnum* species and had no effect on another. The growth rate of recurved bog moss *Sphagnum recurvum* was 12–50% lower in limed pools than in unlimed pools. The effect of liming was especially strong in pools that were also fertilized. The growth rate of feathery bog moss *Sphagnum cuspidatum* was not significantly lower (only 4–8% less) in limed pools than in unlimed pools. In 1991, individual *Sphagnum* plants (cut to 5 cm length) were submerged (30 cm deep) in 4 m^3 pools dug in the bog (number of plants and pools not reported). After 10 days, four treatments were applied: lime with fertilizer, liming only, fertilization only, or none. Limed pools received 80g calcium carbonate. Fertilized pools received 30 g sodium phosphate. The length of all plants was measured after 20 weeks.

A replicated, controlled study in 1994–1995 in a degraded fen meadow in the Netherlands (2) reported that liming typically reduced survival of planted herbs. Three species were planted: carnation sedge *Carex panicea*, tawny sedge *Carex hostiana* and meadow thistle *Cirsium dissectum*. In four of six comparisons, survival after two growing seasons was lower in limed plots (8–20%) than in unlimed plots (15–32%). In one comparison, survival was no different in limed and unlimed plots (72%). In the final comparison, survival was higher in limed plots (88%) than in unlimed plots (80%). After one growing season, lime had little effect on survival (>92% in all plots). In May 1994, twenty 1 m^2 plots were each planted with 15 plants (five of each species). Ten plots were limed (450–510 g/m^2) and ten were not. All plots had been rewetted and were mown every August. Half had been stripped of topsoil. In August 1994 and 1995, survival of all plants was recorded.

A replicated, randomized, paired, controlled, before-and-after study in 2004–2005 in a degraded fen in Sweden (3a) found that liming increased survival and spread of sown fen mosses. After one growing season, moss survival was significantly higher in plots that had been limed (60–93% of plots contained live moss shoots) than in unlimed plots (4–48%). After two growing seasons, moss cover was significantly higher in limed plots (8–34%) than in unlimed plots (<1–10%). In June 2004, ninety-six 9 cm^2 plots were established, in four equal blocks, on rewetted (historically drained) bare peat. Fragments of four fen-characteristic moss species were added (16 fragments of a single species in 9 cm^2 subplots) to 24 plots (625 cm^2). Twelve of these had been limed before planting (1.2 kg/m^2, raising pH from 4.9 to 6.3). Some plots were also covered (with sedge litter or plastic gauze) after planting. Moss survival was assessed after one growing season. Moss cover was visually estimated after two growing seasons.

A replicated, controlled, before-and-after study in a greenhouse in Sweden (3b) found that liming increased the growth rate of planted fen mosses. After five months, shoots of all four planted moss species were longer in limed trays (42–91 mm) than in unlimed trays (29–61 mm). When planted, fragments were 10 mm long. Four trays of peat (19 x 56 cm) were each planted with 160 moss fragments: ten clusters of four fragments, for each of four species. In two of the trays, lime had been mixed into the peat before planting (128 g/tray). All trays were covered with clear plastic lids, kept in controlled light conditions, watered and systematically rearranged every 10 days. After five months, the length of all planted fragments was measured.

A replicated, randomized, controlled, before-and-after study in 2011 in a nursery in Indonesia (4) found that liming typically had no effect on growth of planted tree seedlings. Seedlings of 22 peat swamp tree species were studied. Limed and unlimed seedlings showed similar height growth for 15 species, similar growth of stem diameter for 14 species, and similar increase in dry mass for 19 species. The remaining species showed mixed responses: liming increased growth of some but reduced growth of others. In June 2011, 10 random seedlings of each species were limed (36.8 mg dolomitic lime twice/week) and 10 were not. Seedlings were grown in pots of soil and rice husk, from seed or transplanted from the wild. The duration of the experiment was not reported.

A replicated, randomized, paired, controlled study in 2011–2013 in a historically mined bog in Quebec, Canada (5) found that liming plots sown with vegetation fragments had no effect on vegetation cover. After two years, there was no significant difference between limed and unlimed plots for total vegetation cover (limed: 25%; unlimed: 21%), vascular plant cover (limed: 21%; unlimed: 18%) or bryophyte cover (limed: 4%; unlimed: 3%). In winter 2009/2010, nine pairs of 20 m^2 plots were sown with mixed vegetation fragments from a donor fen. The plots were on a historically mined bog, but the aim of this study was to create a fen because the post-mining peat chemistry was more fen-like than bog-like. In July 2012, dolomitic lime was added to one plot/pair (15 g/m^2). The other plots were not limed. In July 2014, vegetation cover was estimated in six quadrats/plot: vascular plants in three 1 x 1 m quadrats and bryophytes in three 50 x 50 cm quadrats.

(1) Money R.P. (1995) Re-establishment of a *Sphagnum* dominated flora on cut-over lowland raised bogs. Pages 405-422 in B.D. Wheeler, S.C. Shaw, W.J. Fojt & R.A. Robertson (eds.): *Restoration of Temperate Wetlands.* John Wiley and Sons Ltd, Chichester.
(2) van Duren I.C., Strykstra R.J., Grootjans A.P., ter Heerdt G.N.J. & Pegtel D.M. (1998) A multidisciplinary evaluation of restoration measures in a degraded *Cirsio-Molinietum* fen meadow. *Applied Vegetation Science*, 1, 115–130.
(3) Mälson K. & Rydin H. (2007) The regeneration capabilities of bryophytes for rich fen restoration. *Biological Conservation*, 135, 435–442.
(4) Yuwati T.W., Rachmanadi D., Santosa P.B., Rusmana & Graham L.L.B. (2014) Response of peat swamp forest species to macronutrients. Pages 46–63 in: Banjarbaru Forestry Research Unit,

FORDA & L.L.B. Graham (eds.) *Tropical Peat Swamp Forest Silviculture in Central Kalimantan*. Kalimantan Forests and Climate Partnership, Indonesia.
(5) Rochefort L., LeBlanc M.-C., Bérubé V., Hugron S., Boudreau S. & Pouliot R. (2016) Reintroduction of fen plant communities on a degraded minerotrophic peatland. *Canadian Journal of Botany*, 94, 1041–1051.

13.2 Add inorganic fertilizer (before/after planting) Ⓑ Ⓕ Ⓢ

- **Nine studies** examined the effect, on peatland vegetation, of adding inorganic fertilizer to areas planted with peatland plants. Eight studies were in bogs[1,2,3,4,5a,5b,6,8] (two[6,8] being restored as fens). One study was in a tropical peat swamp nursery[7].
- **Survival (2 studies):** Two replicated, randomized, paired, controlled studies in bogs in Canada examined the effect, on plant survival, of adding inorganic fertilizer to areas planted with peatland plants. One study[5a] reported that fertilizer increased survival of two planted tree species. The other study[4] found that fertilizer had no effect on three planted tree species and reduced survival of one.
- **Growth (6 studies):** Five studies (three replicated, randomized, paired, controlled) in bogs in the UK[1], Germany[2] and Canada[4,5a,5b] found that fertilizer typically increased growth of planted mosses[1], herbs[2] or trees[4,5a,5b]. However, for some species[2,4] or in some conditions[1], fertilizer had no effect on growth. One replicated, randomized, controlled, before-and-after study in a nursery in Indonesia[7] found that fertilizer typically had no effect on growth of peat swamp tree seedlings.
- **Cover (3 studies):** Three replicated, randomized, paired, controlled studies examined the effect, on vegetation cover, of fertilizing areas planted with peatland plants. One study in a bog in Canada[8] found that fertilizer increased total vegetation, vascular plant and bryophyte cover. Another study in a bog (being restored as a fen) in Canada[6] found that fertilizer increased sedge cover but had no effect on total vegetation cover, total herb cover or *Sphagnum* moss cover. One study in a bog in New Zealand[3] reported that fertilizer typically increased cover of a sown shrub and rush, but this depended on the chemical in the fertilizer and preparation of the peat.

Background

Inorganic fertilizer can provide nutrients that are in short supply, thereby increasing the initial survival and/or growth rate of introduced plants. Commonly added nutrients include nitrogen (N), phosphorous (P) and/or potassium (K). These are major nutrients that are most commonly limiting for plant growth in peatlands (Rydin & Jeglum 2013). Fertilizer is usually added immediately before or immediately after planting. Note that we do not include studies that only report responses of nurse plants (e.g. Caporn *et al.* 2007).

CAUTION: Peatlands are characterized by low nutrient availability (Rydin & Jeglum 2013). Adding fertilizer might be a short-term solution to encourage initial plant growth, but could lead to undesirable long-term increases in nutrient levels. Some studies involve planting trees, but trees are not natural features of many peatlands.

Related interventions: add fertilizer to peatlands without introducing vegetation (Section 12.10); restoration using multiple interventions, including fertilization (Section 12.1); introduce peatland vegetation – overall effects (Sections 12.16–12.18).

Caporn S., Sen R., Field C., Jones E., Carroll J. & Dise N. (2007) *Consequences of Lime and Fertilizer Application for Moorland Restoration and Carbon Balance*. Moors for the Future Research Report.

Rydin H. & Jeglum J.K. (2013) *The Biology of Peatlands, Second Edition*. Oxford University Press, Oxford.

A controlled, before-and-after study in 1991 in a historically mined raised bog in England, UK (1) found that fertilization increased growth of planted *Sphagnum* mosses in three of four cases. In pools with no other intervention, both *Sphagnum* species grew faster when fertilized than when not fertilized (feathery bog moss *Sphagnum cuspidatum* 30% faster and recurved bog moss *Sphagnum recurvum* 85% faster). Amongst limed pools, only feathery bog moss grew faster (35%) when fertilized. In limed pools, fertilizer had no effect on recurved bog moss growth. In 1991, individual *Sphagnum* plants (cut to 5 cm length) were submerged (30 cm deep) in 4 m^3 pools dug in the bog (number of plants and pools not reported). After 10 days, four treatments were applied: fertilization and liming, fertilization only, liming only, or none. Fertilized pools received 30 g sodium phosphate. Limed pools received 80g calcium carbonate. The length of all plants was measured after 20 weeks.

A replicated, controlled, before-and-after study in 1991–1995 in a historically mined raised bog in Germany (2) found that fertilization increased growth of two of three planted herb species. One year after fertilization, there were more shoots on fertilized than unfertilized sedge *Carex rostrata* (142 vs 45 shoots/plant) and common cottongrass *Eriophorum angustifolium* (6 vs 2 shoots/plant). However, sheathed cottongrass *Eriophorum vaginatum* tussocks had similar diameters whether fertilized (69 cm) or not (70 cm). In 1991, twelve 3 x 35 m plots of bare rewetted peat were planted with the shoots and tussocks (one plant/3 m^2). At this point, plants destined to be fertilized and unfertilized did not differ in shoot number or tussock diameter. In 1994, six of the plots were fertilized (mixture of N, P and K compounds; 100 g/m^2). In 1995, shoot number and tussock diameter were re-measured on each plant.

A replicated, randomized, paired, controlled, before-and-after study in 1998–2000 in a historically mined raised bog in New Zealand (3) reported that fertilization had mixed effects on cover of two sown plant species after 810 days. These results are not based on tests of statistical significance. Fertilizer that included phosphorous increased cover of manuka *Leptospermum scoparium* (or did not reduce it from 100%) in all six cases, but fertilizing only with nitrogen reduced cover in all three cases. Fertilization increased cover of bamboo rush *Sporadanthus ferrugineus* in five of six cases on tilled plots (fertilized: 3–11%; unfertilized 1–2%), but reduced rush cover in all three cases on raised plots (fertilized: 0–6%; unfertilized: 32%). In March 1998, forty-eight 25 m^2 plots were established, in six blocks, on bare rewetted peat (some tilled and some raised). All plots were sown with manuka or bamboo rush seeds. For each plant species, six plots (one random plot/block) received each of four fertilizer treatments: N (100 kg/ha), P (50 kg/ha), N+P, or none. In June 2000, canopy cover of every plant species was estimated. This study also reported the effect of fertilization in unsown plots (see Section 12.10).

A replicated, randomized, paired, controlled study in 1990–2002 in a historically mined bog in Quebec, Canada (4) found that fertilization reduced survival of one planted tree species (no effect on three others) but increased growth of one species (no effect on one other). After three growing seasons, fertilized black spruce *Picea mariana* saplings had lower survival rates (24–65%) than unfertilized saplings (75%). Fertilization did not significantly affect survival of tamarack *Larix laricina*, red maple *Acer rubrum* or poplar *Populus* spp. (fertilized: 1–76%; unfertilized: 16–85%). In the third growing season, tamarack grew more with a low fertilizer dose (shoot length 59–65 cm) than a high dose (46 cm) or no fertilizer (39 cm). Fertilizer did not affect growth of black spruce. In early summer 2000, seedlings of each tree species were planted into bare, slightly drained peat. There were 2–7 single-species blocks/species.

Within each block there were three fertilized plots (mixture of N, P and K compounds; 122.5 g/plant, 245 g/plant or 490 g/plant) and one unfertilized plot. In August 2002, seedling survival was assessed. Terminal shoot length was measured for nine trees (across three evenly spaced quadrats) in each plot.

A replicated, randomized, paired, controlled study in 2004–2005 in a historically mined bog in Quebec, Canada (5a) reported that fertilization increased survival and growth rate of two planted tree species. These results were not tested for statistical significance. After two growing seasons, survival of fertilized tamarack *Larix laricina* was 92–98% (unfertilized: 81%) and of fertilized black spruce *Picea mariana* 58–87% (unfertilized: 55%). As a measure of growth rate, shoot length of fertilized tamarack was 9–33 cm (unfertilized: 1 cm) and of fertilized spruce 4–6 cm (unfertilized: 3 cm). In early June 2004, seedlings of the two tree species were planted into drained bare peat. There were three blocks/species. Within each block, six plots of 150 trees immediately received a random fertilization treatment (commercial or custom-made; see original paper). Three additional plots of 50 trees were not fertilized. In October 2005, seedling survival was assessed. Terminal shoot length was measured for 15 trees (across five randomly placed quadrats) in each plot.

A replicated, randomized, controlled study in 2005–2006 in two historically mined bogs in New Brunswick, Canada (5b) reported that fertilization increased growth of planted trees in 8 of 14 combinations: only when the fertilizer included phosphorous. These results are not based on tests of statistical significance. After two growing seasons, trees fertilized with phosphorous had longer shoots (black spruce *Picea mariana*: 13–23 cm; tamarack *Larix laricina*: 46–65 cm) than unfertilized trees (spruce: 7 cm; tamarack: 2 cm). Trees fertilized only with nitrogen or potassium had similar-length shoots to unfertilized trees (spruce: 4–6 cm; tamarack: 2–3 cm). Saplings were originally planted into bare peat in spring 2001. In spring 2005, four plots of each tree species (one random plot in each of four blocks) received each fertilization treatment: no fertilizer, N (40 g/plant), P (9 g/plant), K (15 g/plant), N+P, N+K, P+K or N+P+K. In October 2006, terminal shoot length of eight trees was measured in each plot.

A replicated, randomized, paired, controlled, before-and-after study in 2004–2006 in a historically mined bog in Quebec, Canada (6) found that fertilizing plots sown with vegetation fragments increased cover of sedges *Carex* spp., but had no effect on cover of other vegetation or plant species richness. Before sowing, all plots were bare peat. After two years, sedge cover was higher in fertilized plots (8%) than unfertilized plots (2%). However, there was no significant difference between fertilized and unfertilized plots for vegetation cover (fertilized: 53%; unfertilized: 47%), *Sphagnum* moss cover (fertilized: 21%; unfertilized: 29%), total herb cover (data not reported) or plant species richness (data not reported). In May–August 2004, vegetation fragments from *Sphagnum*-dominated fens were spread onto five pairs of cleared and levelled 5 x 6 m plots. Note that the aim of this study was to create a fen, as the post-mining peat chemistry was more like a fen than a bog. Five plots (one random plot/pair) were fertilized with rock phosphate (15 g/m^2). The other plots were not fertilized. All plots were mulched with straw. In September 2006, cover of every plant species was estimated in 10–20 quadrats/plot.

A replicated, randomized, controlled, before-and-after study in 2011 in a nursery in Indonesia (7) found that fertilization typically had no effect on growth of planted tree seedlings. Seedlings of 22 peat swamp tree species were studied. For 14–22 species (depending on the chemicals in the fertilizer), fertilized and unfertilized seedlings showed similar height growth. Similarly, fertilization had no significant effect on stem diameter of 16–18 species and dry mass of 19–20 species. The remaining species showed mixed responses: fertilization increased growth of some but reduced growth of others. In June 2011, 10 random seedlings of each species received each fertilizer treatment (36.8 mg of each nutrient twice/week): N, N+P, N+P+K or none. Seedlings were grown in pots of soil and rice husk, from seed or transplanted from the wild. The duration of the experiment was not reported.

A replicated, randomized, paired, controlled study in 2011–2013 in a historically mined bog in Quebec, Canada (8) found that fertilizing plots sown with vegetation fragments increased total vegetation cover, vascular plant cover and bryophyte cover. After two years, fertilized plots had significantly greater cover than unfertilized plots of total vegetation (fertilized: 44%; unfertilized: 21%), vascular plants (fertilized: 35%; unfertilized: 18%) and bryophytes (fertilized: 9%; unfertilized: 3%). Nine pairs of 20 m^2 plots were established on a historically mined bog. The plots had been sown with mixed vegetation fragments from a donor fen in winter 2009/2010. Note that the aim of this study was to create a fen, as the post-mining peat chemistry was more like a fen than a bog. In July 2012, nine plots (one plot/pair) were fertilized with rock phosphate (25 g/m^2). The other plots were not fertilized. In July 2014, vegetation cover was estimated in six quadrats/plot: vascular plants in three 1 x 1 m quadrats and bryophytes in three 50 x 50 cm quadrats.

(1) Money R.P. (1995) Re-establishment of a *Sphagnum* dominated flora on cut-over lowland raised bogs. Pages 405-422 in B.D. Wheeler, S.C. Shaw, W.J. Fojt & R.A. Robertson (eds.): *Restoration of Temperate Wetlands.* John Wiley and Sons Ltd, Chichester.
(2) Sliva J. & Pfadenhauer J. (1999) Restoration of cut-over raised bogs in southern Germany – a comparison of methods. *Applied Vegetation Science*, 2, 137–148.
(3) Schipper L.A., Clarkson B.R., Vojvodic-Cukovic M. & Webster R. (2002) Restoring cut-over restiad peat bogs: a factorial experiment of nutrients, seed and cultivation. *Ecological Engineering*, 19, 29–40.
(4) Bussières J., Boudreau S. & Rochefort L. (2008) Establishing trees on cut-over peatlands in eastern Canada. *Mires and Peat*, 3, Article 10.
(5) Caisse G., Boudreau S., Munson A.D. & Rochefort L. (2008) Fertiliser addition is important for tree growth on cut-over peatlands in eastern Canada. *Mires and Peat*, 3, Article 11.
(6) Graf, M.D. & Rochefort, L. (2008) Techniques for restoring fen vegetation on cut-away peatlands in North America. *Applied Vegetation Science*, 11, 521–528.
(7) Yuwati T.W., Rachmanadi D., Santosa P.B., Rusmana & Graham L.L.B. (2014) Response of peat swamp forest species to macronutrients. Pages 46–63 in: Banjarbaru Forestry Research Unit, FORDA & L.L.B. Graham (eds.) *Tropical Peat Swamp Forest Silviculture in Central Kalimantan*. Kalimantan Forests and Climate Partnership, Indonesia.
(8) Rochefort L., LeBlanc M.-C., Bérubé V., Hugron S., Boudreau S. & Pouliot R. (2016) Reintroduction of fen plant communities on a degraded minerotrophic peatland. *Canadian Journal of Botany*, 94, 1041–1051.

13.3 Add organic fertilizer (before/after planting)

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect, on peatland vegetation, of adding organic fertilizer to areas planted with peatland plants.

Background

Organic fertilizer (i.e. remains or waste products of living organisms) can provide nutrients that are in short supply, thereby increasing the initial survival and/or growth rate of introduced plants. Once successfully established, these plants are more likely to survive long term. Fertilizer would usually be added into the peat immediately before or after planting.

Related interventions: add inorganic fertilizer to complement planting (Section 13.2); introduce peatland vegetation – overall effects (Sections 12.16–12.18).

13.4 Cover peatland with organic mulch (after planting) Ⓑ Ⓕ Ⓢ

- **Twelve studies** examined the effect, on peatland vegetation, of adding organic mulch after planting peatland plants. Nine studies were in bogs[1,2,4a,4b,4c,4d,5,8a,8b] (one[5] being restored as a fen). Two studies were in fens[6,7]. One was in a tropical peat swamp[3].
- **Germination (1 study):** One replicated, controlled, before-and-after study in a bog in Germany[2] found that mulching after sowing seeds increased germination rates for two species (a grass and a shrub), but had no effect on three other herb species.
- **Survival (3 studies):** Two replicated, paired, controlled studies in a fen in Sweden[6] and a bog in the USA[7] reported that mulching increased survival of planted vegetation (mosses[6,7] or sedges[7]). One replicated, paired, controlled study in Indonesia[3] reported that mulching with oil palm fruits reduced survival of planted peat swamp tree seedlings.
- **Growth (1 study):** One replicated, randomized, paired, controlled, before-and-after study in a fen in the USA[7] reported that mulching increased growth of transplanted sedges.
- **Cover (9 studies):** Six studies (including four replicated, randomized, paired, controlled, before-and-after) in bogs in Canada[1,4a,4b,4d] and the USA[4c] and a fen in Sweden[6] found that mulching after planting increased vegetation cover (specifically total vegetation[4a,4b,4c,4d], total mosses/bryophytes[4a,4b,4d,6], *Sphagnum* mosses[1,4a,4b,4d] or vascular plants[4a,4d] after 1–3 growing seasons). Three replicated, randomized, paired, controlled, before-and-after studies in degraded bogs in Canada[5,8a,8b] found that mulching after planting had no effect on vegetation cover (*Sphagnum* mosses[8a,8b] or fen-characteristic plants[5]).

Background

Introduced peatland vegetation may be killed by hot, dry conditions on bare peat surfaces (e.g. Sagot & Rochefort 1996). Organic mulches (e.g. straw, grass cuttings, heather brash or shrub roots) can be placed on the peatland surface after plants have been introduced to stabilize temperatures and humidity, and offer some shading. This creates more hospitable environment for establishment and growth of introduced vegetation (Price *et al.* 1998). Typically, mulch is applied sparsely enough that some light can still reach the peat surface. CAUTION: Mulches may contain seeds of undesirable plants. It may be necessary to sterilize the material before use.

Related interventions: add mulch to peatlands without introducing vegetation (Section 12.11); add other covers such as mats, sheets or screens after planting (Section 13.5); introduce peatland vegetation – overall effects (Sections 12.16–12.18).

Price J., Rochefort L. & Quinty F. (1998) Energy and moisture considerations on cutover peatlands: surface microtopography, mulch cover and *Sphagnum* regeneration. *Ecological Engineering*, 10, 293–312.

Sagot C. & Rochefort L. (1996) Tolérance des sphaignes à la dessiccation (Tolerance of *Sphagnum* mosses to desiccation; in French). *Cryptogamie, Bryology-Lichénologie*, 17, 171–183.

A replicated, randomized, paired, controlled, before-and-after study in 1995–1996 in a historically mined raised bog in Quebec, Canada (1) found that mulching plots sown with *Sphagnum*-dominated vegetation fragments increased *Sphagnum* moss cover. After 1–2 growing seasons, plots mulched with straw after adding the vegetation fragments had significantly higher *Sphagnum* cover (1–5%) than plots that were not mulched (<0.5%). In May 1995, 24 bare peat plots (15 x 15 m, in three blocks of eight) were sown with vegetation fragments (mostly *Sphagnum* moss) from the surface of a nearby bog. Twelve of the plots (four random plots/block) were mulched with straw after sowing (2,250 kg/ha). All plots had been rewetted, and the surface of some was roughened. In June and September 1996, *Sphagnum* cover was estimated in 36–72 quadrats/plot, each 25 x 25 cm.

A replicated, controlled, before-and-after study in 1993–1995 in a historically mined raised bog in Germany (2) found that mulching plots sown with herb or shrub seeds increased germination for two of five species. For purple moor grass *Molinia caerulea* and heather *Calluna vulgaris*, mulched plots contained more seedlings after 1–2 years than unmulched plots (25–45 vs 1–8 seedlings/400 cm^2). For three other herb species, mulched and unmulched plots contained a similar number of seedlings (0–10 vs 0–8 seedlings/400 cm^2). In autumn 1993, seeds of five plant species were spread onto 1 m^2 plots of bare rewetted peat (10 plots/species, 40–48 seeds/400 cm^2). Five plots/species were mulched with leaves or heather branches, whilst five were not mulched. Mulch was removed and seedlings counted in summer 1994 (two plots/treatment) and 1995 (three plots/treatment).

A replicated, paired, controlled study in 1999–2000 in a degraded peat swamp in Malaysia (3) reported that mulching with oil palm fruits reduced the survival of planted tree seedlings. No statistical tests were carried out. After 14 months, 50% of mulched seedlings had survived, compared to 83% of seedlings that were not mulched. The mulch attracted wild boars (which damaged the seedlings) and produced a hot vapour (which may have dried the seedlings). In June 1999, thirty-six plots in a degraded, open peat swamp were planted with peat swamp trees (16 seedlings/plot). There were three pairs of plots for each of six tree species. Eighteen plots (one plot/pair) were mulched with fresh oil palm fruit skins. The other plots were not mulched. All plots were cleared of vegetation before planting and the planting holes were fertilized. Survival was recorded in August 2000.

A replicated, randomized, paired, controlled, before-and-after study in 1994–1996 in a historically mined bog in Quebec, Canada (4a) reported that mulching plots sown with vegetation fragments increased vegetation cover. These results were not tested for statistical significance. After three growing seasons, total vegetation cover was 17–24% in mulched plots but 3–5% in unmulched plots. This included *Sphagnum* moss (mulched: 0–7%; unmulched: <1%), other moss (mulched: 2–13%; unmulched: 1–2%) and vascular plants (mulched: 4–15%; unmulched: 1–4%). In early 1994, mixed plant material was collected from a natural bog and spread onto 12 pairs of plots (each 3 x 15 m) of bare rewetted peat. Then, one random plot in each pair was mulched with straw (1,500 kg/ha). The other plots were not mulched. In 1994 and 1996, vegetation cover was estimated within quadrats in each plot (details not reported).

A replicated, paired, controlled, before-and-after study in 1994–1996 in a historically mined bog in Quebec, Canada (4b) found that amongst plots sown with vegetation fragments, mulching with straw increased total vegetation and moss cover, but shrub root mulch had no effect. After three growing seasons, plots mulched with straw had significantly higher cover than unmulched plots of total vegetation (mulched: 20%; unmulched: 3%), *Sphagnum* moss (mulched: 1%; unmulched: 0%) and other moss (mulched: 16%; unmulched: 2%). Cover of vascular plants was similar (<1%) in mulched and unmulched plots. In contrast, all vegetation groups had similar cover in plots covered with shrub roots and the unmulched plots (total: 4%; *Sphagnum* moss: <1%; other moss: 3%; vascular plants: <1%). In spring 1994, plant material was scraped from the surface of a natural bog and spread onto plots of bare rewetted peat. There were nine 9 m^2 plots, arranged in three blocks of three. One plot/block was then covered with a straw mulch (1,500 kg/ha), one loosely covered with shrub roots (20% cover) and one left uncovered. Vegetation cover was estimated in 1994 and 1996 (details not reported).

A replicated, randomized, paired, controlled, before-and-after study in 1997–1999 in a historically mined bog in Minnesota, USA (4c) found that mulching plots sown with vegetation fragments increased total vegetation and *Sphagnum* moss cover. After two growing seasons, mulched plots had significantly greater cover than unmulched plots of total vegetation (54–77% vs 1–3%) and *Sphagnum* moss (51–73% vs <1%). Mulched and unmulched plots had similar cover of other mosses (<1%) and vascular plants (1–3%). In 1997–1998, vegetation was scraped from the surface of natural bogs and spread onto plots of bare peat. There were forty-eight 1.5 x 1.5 m plots, arranged in six blocks of eight. Four random plots/block were mulched with straw (3,000 kg/ha). The other plots were not mulched. Four plots/block were also planted with sedges *Carex oligosperma* before adding vegetation fragments. In October 1999, vegetation cover was visually estimated in four 25 x 25 cm quadrats/plot.

A replicated, controlled, before-and-after study in 1993–1996 in a historically mined bog in Quebec, Canada (4d) reported that mulching plots sown with vegetation fragments increased vegetation cover. These results were not tested for statistical significance. After three growing seasons, plots mulched with straw had total vegetation cover of 3–11% (vs 2% in unmulched plots), total moss cover of 2–6% (unmulched: <1%), *Sphagnum* moss cover of 1–4% (unmulched: <1%) and vascular plant cover of 1–2% (unmulched: <1%). Amongst mulched plots, vegetation cover (of all groups) was higher when more mulch was added. However, cover was similar in plots mulched in autumn or spring (see original paper). In autumn 1993, vegetation was scraped from the surface of a natural bog and spread onto a ploughed, bare peat site. Within this site, 10 x 10 m plots were mulched with straw immediately or in the following spring: 750, 1,500 or 3,000 kg/ha (number of plots not reported). Some additional plots were not mulched. In autumn 1996, vegetation cover was visually estimated in fourteen 25 x 25 cm quadrats/plot.

A replicated, randomized, paired, controlled, before-and-after study in 2001–2002 in a historically mined bog in Quebec, Canada (5) found that mulching plots sown with vegetation fragments increased the number of fen-characteristic plant species but had no effect on fen-characteristic plant cover. Note that the aim of this study was to create a fen, as the post-mining peat chemistry was more like a fen than a bog. Before sowing, no vegetation was present. After two growing seasons, there were more plant species typical of local fens in mulched plots (13–15 species) than unmulched plots (10–12 species). Fen plant cover did not differ between mulched (21–30%) and unmulched plots (23–32%). Mulching had similar effects in additional plots that were not sown with vegetation fragments (see Section 12.11). In spring 2001, soil and vegetation from nearby moss or grass-dominated fens was spread onto thirty-six 5 x 5 m plots (arranged in three equal blocks). Eighteen plots (six random plots/block) were mulched with straw (1,500 kg/ha). The other plots were not mulched. All plots had previously been rewetted, raked and fertilised. In August 2002, cover of every plant species was estimated in ten 30 x 30 cm quadrats/plot.

A replicated, randomized, paired, controlled, before-and-after study in 2004–2005 in a degraded fen in Sweden (6) reported that mulching with sedge litter increased survival and growth of planted moss fragments, but only when plots were not limed. These results are not based on tests of statistical significance. Amongst unlimed plots, moss survival after one growing season was higher in mulched plots (13% of plots contained live moss) than in unmulched plots (4%). Moss cover after two growing seasons was higher in mulched plots (3–7%) than unmulched plots (≤1%). However, amongst limed plots, mulching had no effect on survival (mulched: 62%; unmulched: 60%) or growth (mulched: 13–24%; unmulched: 8–28%). In June 2004, fragments of four fen-characteristic moss species were added (16 fragments of a single species in 9 cm^2 subplots) to 16 plots (625 cm^2) of bare rewetted peat. Eight plots were then sparsely mulched with sedge *Carex lasiocarpa* litter. The other eight plots were not mulched. Eight plots were also limed before planting. Moss survival was assessed after one growing season and moss cover visually estimated after two.

A replicated, paired, controlled, before-and-after study in 2007–2010 in three degraded fens in Colorado, USA (7) found that mulching increased survival and growth of transplanted water sedge *Carex aquatilis*, and survival of sown moss fragments. For transplanted sedges, survival over three years was higher in mulched plots than in unmulched plots (55 vs 35%). The same was true for growth (30–82 vs 7–81 stems/plot). No moss survived on unmulched plots. Under mulch, Russow's bog moss *Sphagnum russowii* survived in one of three sites reaching 19% cover after three years. Under mulch, haircap moss *Polytrichum strictum* survived in all three sites reaching 3–11% cover after three years. In July 2007, thirty-six plots were established (in six blocks of six) on bare peat. Twelve plots (two plots/block) received each planting treatment: sedges (18 single stems/plot), mosses (mixed *Sphagnum* and haircap moss fragments; 4.4 L/plot) or sedges and mosses. Half of the plots were mulched with straw (immediately) and shredded aspen (after one year). The other plots were not mulched. In summer 2010, sedge survival, sedge stem number and moss cover were recorded.

A replicated, randomized, paired, controlled, before-and-after study in 2007–2010 in two historically disturbed bogs in Ontario, Canada (8a) found that mulching plots sown with *Sphagnum* moss fragments had no effect on bryophyte cover. After three years, *Sphagnum* cover did not differ significantly between treatments (sedge mulch: 33%; coconut mulch: 42%; straw mulch: 52%; no mulch: 38%). There was also no difference in total bryophyte cover between treatments (mulch: 66–76%; no mulch: 68%). In August 2007, fragments of rusty bog moss *Sphagnum fuscum* and flat-topped bog moss *Sphagnum fallax* were spread onto 24 bare peat plots (each 2 x 2 m, arranged in six blocks of four). Six plots (one random plot/block) received each mulch treatment: none, sedge cuttings, coconut fibre or straw. All plots were also fertilized with rock phosphate. In August 2010, moss cover was estimated in six random 12.5 x 12.5 cm subplots within each plot.

A replicated, randomized, paired, controlled, before-and-after study in 2007–2010 in two historically disturbed bogs in Ontario, Canada (8b) found that mulching plots sown with *Sphagnum* moss fragments had no effect on bryophyte cover. After three years, *Sphagnum* cover did not significantly differ between mulched plots (9–

17%) and unmulched plots (3–7%). There was also no difference in total bryophyte cover between mulched plots (29–51%) and unmulched plots (24–31%). In May 2007, fragments of rusty bog moss *Sphagnum fuscum* and flat-topped bog moss *Sphagnum fallax* were spread onto 24 bare peat plots (each 1 m^2, arranged in four blocks of six). Twelve plots (three random plots/block) were then mulched with straw. The other 12 plots were not mulched. All plots were fertilized with rock phosphate. In August 2010, moss cover was estimated in six random 12.5 x 12.5 cm subplots within each plot.

(1) Price J., Rochefort L. & Quinty F. (1998) Energy and moisture considerations on cutover peatlands: surface microtopography, mulch cover and *Sphagnum* regeneration. *Ecological Engineering*, 10, 293–312.

(2) Sliva J. & Pfadenhauer J. (1999) Restoration of cut-over raised bogs in southern Germany – a comparison of methods. *Applied Vegetation Science*, 2, 137–148.

(3) Ismail P., Shamsudin I., Nik Muhamad N.M. & Faridah Hanum I. (2001) *Rehabilitation of grassland areas in peat swamp forests in Peninsular Malaysia*. Proceedings of the Asian Wetland Symposium 2001, Penang, Malaysia, 57– 64.

(4) Rochefort L., Quinty F., Campeau S., Johnson K. & Malterer T. (2003) North American approach to the restoration of *Sphagnum* dominated peatlands. *Wetlands Ecology and Management*, 11, 3–20.

(5) Cobbaert D., Rochefort L. & Price J.S. (2004) Experimental restoration of a fen plant community after peat mining. *Applied Vegetation Science*, 7, 209–220.

(6) Mälson K. & Rydin H. (2007) The regeneration capabilities of bryophytes for rich fen restoration. *Biological Conservation*, 135, 435–442.

(7) Chimner R.A. (2011) Restoring sedges and mosses into frost heaving iron fens, San Juan Mountains, Colorado. *Mires and Peat*, 8, Article 7.

(8) Corson A. & Campbell D. (2013) Testing protocols to restore disturbed *Sphagnum*-dominated peatlands in the Hudson Bay Lowland. *Wetlands*, 33, 291–299.

13.5 Cover peatland with something other than mulch (after planting) Ⓑ Ⓕ Ⓢ

- **Eight studies** examined the effect, on peatland vegetation, of adding covers (other than mulch) after planting peatland plants. Five studies involved bog plants[1,2,3,4,6], two involved fen plants[5a,5b] and one involved peat swamp plants[7]. Two of the studies were in greenhouses or nurseries[1,7].
- **Germination (1 study):** One replicated, controlled, before-and-after study in a bog in Germany[3] reported mixed effects of fleece and fibre mats on germination of sown herb and shrub seeds (positive or no effect, depending on species).
- **Survival (2 studies):** Two replicated, randomized, controlled studies examined the effect, on plant survival, of covering planted areas. One study in a fen in Sweden[5a] reported that shading with plastic mesh increased survival of planted mosses. One study in a nursery in Indonesia[7] reported that shading with plastic mesh typically had no effect on survival of peat swamp tree species, but increased survival of some.
- **Growth (3 studies):** Three replicated, randomized, controlled, before-and-after studies examined the effect, on plant growth, of covering planted areas. One study in a greenhouse in Switzerland[1] found that covering planted *Sphagnum* mosses with transparent plastic sheets or shading mesh increased their growth. One study in a fen in Sweden[5b] found that shading with plastic mesh reduced growth of planted fen mosses. One study in a nursery in Indonesia[7] reported that seedlings shaded with plastic mesh grew taller and thinner than unshaded seedlings.
- **Cover (4 studies):** Two replicated, paired studies in a fen in Sweden[5a] and a bog in Australia[6] reported that shading plots with plastic mesh increased cover of planted mosses. One study in a bog in Canada[2] found that covering sown plots with plastic mesh, but not transparent plastic

sheets, increased the number of *Sphagnum* moss shoots. Another study in a bog in Canada[4] reported that shading sown plots with plastic mesh had no effect on cover of vegetation overall, vascular plants, *Sphagnum* or other moss.

Background

Introduced peatland vegetation may be killed by hot, dry and bright conditions on bare peat surfaces (e.g. Harley *et al.* 1989; Sagot & Rochefort 1996). Covers (e.g. plastic sheets, fleece or geojute fibre mats) can physically stabilize the peat surface, maintain more constant temperatures and humidity, and offer some shading. This creates a more hospitable environment for establishment and growth of introduced vegetation (Price *et al.* 1998).

This section considers covers that may be placed on peatlands as sheets to control light and/or moisture levels. The precise effect may vary depending on the material and the height above the peatland. We use the term *mesh* to describe all net-like covers with lots of small holes, used primarily to shade the peatland surface. This includes shade screens, shade cloths, gauze and netting.

Related interventions: cover peatlands without introducing vegetation (Section 12.12); add organic mulch after planting (Section 13.4); introduce peatland vegetation – overall effects (Sections 12.16–12.18).

Harley P.C., Tenhunen J.D., Murray K.J. & Beyers J. (1989) Irradiance and temperature effects on photosynthesis of tussock tundra *Sphagnum* mosses from the foothills of the Philip Smith Mountains, Alaska. *Oecologia*, 79, 251–259.

Price J., Rochefort L. & Quinty F. (1998) Energy and moisture considerations on cutover peatlands: surface microtopography, mulch cover and *Sphagnum* regeneration. *Ecological Engineering*, 10, 293–312.

Sagot C. & Rochefort L. (1996) Tolérance des sphaignes à la dessiccation (Tolerance of *Sphagnum* mosses to desiccation; in French). *Crytogamie, Bryology-Lichénologie*, 17, 171–183.

A replicated, randomized, controlled, before-and-after study in a greenhouse in Switzerland (1) found that planted *Sphagnum* moss grew longer, thinner shoots in pots covered with plastic sheets or mesh than in uncovered pots. Over 16 weeks, *Sphagnum* increased in length significantly more in covered pots (plastic sheet: <10–90 mm; plastic mesh: 6–68 mm; uncovered: 3–46 mm increase). However, neither cover significantly affected *Sphagnum* mass growth (sheet: 0.8–5.5; mesh: 0.1–2.2; uncovered: 0.3–4.2 proportional increase). In May (year not reported), 90 pots of peat were planted with flat-topped bog moss *Sphagnum fallax*: twelve 3 cm fragments/pot. Thirty pots were then covered with clear green plastic (with 1 cm diameter holes covering about 5% of the surface area), 30 were shaded with plastic mesh (blocking 80% of incoming light), and 30 left uncovered. All pots were kept in random positions in a greenhouse with controlled temperature, humidity, light and water. After 16 weeks, length and dry mass of all moss fragments were measured.

A replicated, randomized, paired, controlled, before-and-after study in 1993–1994 in a historically mined bog in Quebec, Canada (2) found that amongst plots sown with *Sphagnum*-dominated vegetation fragments, those shaded with plastic mesh contained more *Sphagnum* moss shoots than those covered with transparent plastic sheets or not covered. After two growing seasons, there were significantly more *Sphagnum* shoots in plots covered with plastic mesh (140–510/m^2) than plots covered with plastic sheets (10–30/m^2) or uncovered plots (65–70/m^2). Further, the number of *Sphagnum* shoots increased over the second growing season in shaded plots, but decreased in the other plots. Covers had a similar effect on all focal *Sphagnum* species. In spring 1993, twelve plots (three blocks of four) of slightly drained bare peat were sown with vegetation fragments (mostly *Sphagnum* moss; 250 fragments/m^2). In each plot, subplots received fragments dominated by one of five single *Sphagnum* species. Three plots (one random plot/block) received each cover treatment: no cover, Agrinet™ 40% plastic mesh, Agrinet™ 60% plastic mesh, or a transparent polythene sheet (with 3 cm diameter holes cut 30 cm apart). Covers were 15–20 cm above the bog surface. In autumn 1993 and 1994, *Sphagnum* shoots were counted in ten 30 x 30 cm quadrats/plot.

A replicated, controlled, before-and-after study in 1993–1995 in a historically mined raised bog in Germany (3) reported that covering plots with fleece or fibre mats did not affect germination of three of five sown species but increased germination of the other two. These results are not based on tests of statistical significance. For three herb species, there were a similar number of seedlings after 1–2 years in covered plots (0–11 seedlings/400 cm^2) and uncovered plots (0–10 seedlings/400 cm^2). In contrast, for one herb and one shrub species, there were 14–27 seedlings/400 cm^2 in covered plots but only 1–8 seedlings/400 cm^2 in uncovered plots. Fleece and fibre mat had similar effects on seedling number (see original paper). Covers had no effect on germination in additional plots that were not sown (see Section 12.12). In autumn 1993, seeds of five plant species were spread onto 1 m^2 plots of bare rewetted peat (15 plots/species; 40–48 seeds/400 cm^2). Five plots/species were covered with synthetic fleece, five with wide-meshed jute fibre mat, and five were not covered. Covers were removed and seedlings counted in summer 1994 (two plots/treatment) and 1995 (three plots/treatment).

A replicated, paired, controlled, before-and-after study in 1994–1996 in a historically mined bog in Quebec, Canada (4) reported that shading plots sown with vegetation fragments had no effect on vegetation cover. These results are not based on tests of statistical significance. Plots were initially rewetted bare peat. After three growing seasons, shaded and unshaded plots both had 3–4% total vegetation cover, <1% *Sphagnum* moss cover, 2% other moss cover and <1% vascular plant cover. In spring 1994, the moss layer was scraped from the surface of a natural bog and spread onto three pairs of bare peat plots (each 9 m^2). Then, one plot in each pair was shaded with plastic mesh (Agrinet™ 57%). The other plots were not shaded. In 1994 and 1996, vegetation cover was estimated in each plot (details not reported).

A replicated, randomized, paired, controlled, before-and-after study in 2004–2005 in a degraded fen in Sweden (5a) reported that shading plots with plastic mesh increased survival and spread of planted moss fragments. These results are not based on tests of statistical significance. After one growing season, moss survival was 48–93% in shaded plots but only 4–60% in unshaded plots. After two growing seasons the pattern was similar: moss survival was 5–34% in shaded plots but <1–28% in unshaded plots. In June 2004, fragments of four fen-characteristic moss species were added (16 fragments of a single species in 9 cm^2 subplots) to 16 plots (625 cm^2) of bare rewetted peat. Eight plots were then shaded with plastic horticultural mesh (blocking 15% of incoming light). The other eight plots were not shaded. Some plots were also limed before planting. Moss survival was assessed after one growing season. Moss cover was visually estimated after two growing seasons.

A replicated, randomized, controlled, before-and-after study in a degraded fen in Sweden (5b) found that shading trays with plastic mesh reduced growth of one of two planted moss species, but did not affect growth of the other. After four months, shoots of intermediate hook moss *Scorpidium cossonii* were lighter in shaded trays (5–10 mg dry mass) than in unshaded trays (7–14 mg). In contrast, the mass of starry feather moss *Campylium stellatum* shoots was similar in shaded (4–5 mg) and unshaded trays (3–6 mg). When planted, shoots weighed approximately 1 mg. Six 33 x 33 cm trays of limed peat were set up, floating in a drainage ditch. Three of the trays were shaded with plastic mesh (blocking 45% of incoming light). Each tray was planted with 32 random fragments of each moss species, in single-species clusters. A subsample of fragments was dried and weighed before planting. After four months, all planted fragments were collected, rinsed, dried and weighed.

A replicated, paired, before-and-after study in 2003–2007 in a fire-damaged bog in Australia (6) reported that amongst plots planted with sods of *Sphagnum* moss, those shaded with a horizontal plastic mesh developed greater *Sphagnum* cover than those shaded by a vertical mesh. These results were not tested for statistical significance. Immediately before planting, *Sphagnum* cover was 3% on average. Forty months after planting, horizontally shaded plots had 21% *Sphagnum* cover, compared to 11% in vertically shaded plots. In January 2003, the focal bog was burned by a wild fire. In October 2003, five pairs of plots (3 x 15 m) were planted with sods (20 x 20 x 30 cm) of mixed *Sphagnum* moss species. All sods were fertilized. In each pair, one plot was covered with plastic mesh (blocking 70% of incoming light) and one was shaded with a vertical mesh fence (1.6 m high). *Sphagnum* cover was estimated in 0.25 m^2 quadrats: five in the bog in October 2003, and 1–2/plot in March 2007.

A replicated, randomized, controlled, before-and-after study in 2011 in a nursery in Indonesia (7) reported that shading with plastic mesh typically had no effect on survival of planted tree seedlings, but that shaded seedlings grew taller and thinner than unshaded seedlings. These results are not based on tests of statistical significance. Shading had no effect on survival to four months for 10 of 20 species (100% whether shaded or not) but increased survival for 8 of 20 species (shaded: 80–100%; full sun: 60–90%). Shaded seedlings typically grew taller (18 of 19 species) but had thinner stems (12 of 19 species) than seedlings grown in full sun (see original paper for data). In August 2011, 10 random seedlings of each species received each shade treatment: 75%, 50% or none (full sun). Seedlings were grown in pots from seed or transplanted from the wild. Shade was created with one or two layers of plastic mesh, each layer blocking 50% of incoming light. Measurements were taken before (August) and after four months of shading (December).

(1) Buttler A., Grosvernier P. & Matthey Y. (1998) Development of *Sphagnum fallax* diaspores on bare peat with implications for the restoration of cut-over bogs. *Journal of Applied Ecology*, 35, 800–810.

(2) Rochefort L. & Bastien D.F. (1998) Réintroduction de sphaignes dans une tourbière exploitée: évaluation de divers moyens de protection contre la dessiccation (Reintroduction of *Sphagnum* to an exploited bog: evaluation of various methods for protection against desiccation; in French). *Écoscience*, 5, 117–127.

(3) Sliva J. & Pfadenhauer J. (1999) Restoration of cut-over raised bogs in southern Germany – a comparison of methods. *Applied Vegetation Science*, 2, 137–148.

(4) Rochefort L., Quinty F., Campeau S., Johnson K. & Malterer T. (2003) North American approach to the restoration of *Sphagnum* dominated peatlands. *Wetlands Ecology and Management*, 11, 3–20.

(5) Mälson K. & Rydin H. (2007) The regeneration capabilities of bryophytes for rich fen restoration. *Biological Conservation*, 135, 435–442.

(6) Whinam J., Hope G., Good R. & Wright G. (2010) Post-fire experimental trials of vegetation restoration techniques in the peatlands of Namadgi (ACT) and Kosciuszko National Parks (NSW), Australia. Pages 363–379 in: S.G. Haberle, J. Stevenson & M. Prebble (eds.) *Altered ecologies: fire, climate and human influence on terrestrial landscapes*. Terra Australis 32, Australian National University e-press, Canberra, Australia.

(7) Rusmana, Rachmanadi D., Santosa P.B., Yuwati T.W. & Graham L.L.B. (2014) Response of peat swamp forest species to light intensity. Pages 2–14 in: Banjarbaru Forestry Research Unit, FORDA & L.L.B. Graham (eds.) *Tropical Peat Swamp Forest Silviculture in Central Kalimantan*. Kalimantan Forests and Climate Partnership, Indonesia.

13.6 Introduce nurse plants (to aid focal peatland plants) Ⓑ Ⓕ Ⓢ

- **Three studies** examined the effect, on peatland vegetation, of introducing nurse plants to aid focal peatland plants. Two studies were in bogs[2,3]. One was in a tropical peat swamp[1].
- **Survival (1 study):** One replicated, paired, controlled study in Malaysia[1] reported that planting nurse trees had no effect on survival of planted peat swamp tree seedlings (six species).
- **Cover (2 studies):** Two replicated, randomized, paired, controlled, before-and-after studies in bogs in the USA[2] and Canada[3] found that planting nurse herbs had no effect on cover, after 2–3 years, of other planted vegetation (mosses/bryophytes[2,3], vascular plants[2] or total cover[2]).

Background

Nurse or companion plants can be planted alongside focal plants to help the focal plants establish (Padilla & Pugnaire 2006). Nurse plants can bind together loose peat and reduce harsh environmental conditions (temperature fluctuations, desiccation and strong sunlight). Nurse plants must be able to tolerate these conditions, but invasive species (that spread easily) and species that may outcompete focal plants (physically or chemically) should be avoided. Instead, it is expected that nurse plants are eventually outcompeted *by* focal plants.

Herbs are common nurse plants in temperate peatlands, creating shelter for mosses. They may be directly planted or added as seed (the latter being a more efficient way to cover larger areas). Haircap moss *Polytrichum strictum* may act as a nurse plant for *Sphagnum* moss and vascular plant seedlings (based on a correlative study of natural colonization, and an experiment using non-peatland species; Groeneveld *et al*. 2007). In tropical peat swamps, light-tolerant trees can be used to shelter shade-loving trees.

To be included as evidence in this section, studies must have deliberately introduced nurse plants before planting focal peatland vegetation. Studies that planted peatland vegetation into *existing* nurse vegetation (e.g. Sliva & Pfadenhauer 1999), or examined natural colonization of nurse vegetation (e.g. Groeneveld *et al*. 2007) are not included.

Related interventions: introduce nurse plants as one of many interventions e.g. lime/seed/fertilizer/mulch (Section 12.1); introduce nurse plants but not peatland vegetation (Section 12.14); introduce peatland vegetation (Sections 12.16–12.18).

Groeneveld E.V.G., Masse A. & Rochefort L. (2007) *Polytrichum strictum* as a nurse-plant in peatland restoration. *Restoration Ecology*, 15, 709–719.

Padilla F.M. & Pugnaire F.I. (2006) The role of nurse plants in the restoration of degraded environments. *Frontiers in Ecology and the Environment*, 4, 196–202.

Sliva J. & Pfadenhauer J. (1999) Restoration of cut-over raised bogs in southern Germany – a comparison of methods. *Applied Vegetation Science*, 2, 137–148.

A replicated, paired, controlled study in 1999–2000 in a degraded peat swamp forest in Malaysia (1) reported that planting nurse trees had no effect on the survival of focal planted tree seedlings. No statistical tests were carried out. Six different tree species were planted. After 14 months, 82% of seedlings had survived in plots with nurse trees, compared to 83% of seedlings in plots without nurse trees (data not reported separately for each species). In June 1999, thirty-six plots in a degraded, open peat swamp were planted with peat swamp trees (16 seedlings/plot). There were three pairs of plots for each of six tree species. Eighteen plots (one plot/pair) were also planted with 2 m tall *Hopea odorata* as nurse trees. The other plots contained no additional trees. All plots were cleared of vegetation before planting and the planting holes were fertilized. Survival was recorded in August 2000.

A replicated, randomized, paired, controlled, before-and-after study in 1997–1999 in a historically mined bog in Minnesota, USA (2) found that planting nurse herbs before adding vegetation fragments had no effect on vegetation cover. After two growing seasons, plots with and without nurse herbs had similar cover of total vegetation (3–77% vs 1–71%), *Sphagnum* mosses (0–68% vs 0–73%), other mosses (<1% with or without nurse herbs) and vascular plants (1–3% with or without nurse herbs). In 1997–1998, forty-eight 1.5 x 1.5 m plots were established, in six blocks of eight. Twenty-four plots (four random plots/block) were planted with 16 fewseed sedge *Carex oligosperma* plants. The other 24 plots were left as bare peat. Then, all plots were sown with fresh vegetation fragments from the surface of natural bogs. Some plots with and without nurse sedges were also mulched with straw. In October 1999, vegetation cover was visually estimated in four 25 x 25 cm quadrats/plot.

A replicated, randomized, paired, controlled, before-and-after study in 2007–2010 in two historically disturbed bogs in Ontario, Canada (3) found that planting nurse herbs before sowing *Sphagnum* moss fragments had no effect on bryophyte cover. After three years, *Sphagnum* cover did not differ significantly between plots with nurse herbs (low density: 52%; high density: 51%) and without (38%). There was also no difference in total bryophyte cover between treatments (with nurse plants: 72–76%; without: 68%). In August 2007, eighteen 2 x 2 m plots were established, in six blocks of three, on historically disturbed bogs. Twelve plots (two random plots/block) were planted with cottongrass *Eriophorum vaginatum* tussocks as nurse plants: six at low density (50 cm apart) and six at high density (25 cm apart). The other six plots were left as bare peat. Then, all plots received fresh fragments of rusty bog moss *Sphagnum fuscum* and flat-topped bog moss *Sphagnum fallax*, and 30 g/m^2 rock phosphate fertilizer. In August 2010, moss cover was estimated by eye in six random 12.5 x 12.5 cm subplots within each plot.

(1) Ismail P., Shamsudin I., Nik Muhamad N.M. & Faridah Hanum I. (2001) *Rehabilitation of grassland areas in peat swamp forests in Peninsular Malaysia*. Proceedings of the Asian Wetland Symposium 2001, Penang, Malaysia, 57– 64.
(2) Rochefort L., Quinty F., Campeau S., Johnson K. & Malterer T. (2003) North American approach to the restoration of *Sphagnum* dominated peatlands. *Wetlands Ecology and Management*, 11, 3–20.
(3) Corson A. & Campbell D. (2013) Testing protocols to restore disturbed *Sphagnum*-dominated peatlands in the Hudson Bay Lowland. *Wetlands*, 33, 291–299.

13.7 Rewet peatland (before/after planting)

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect, on peatland vegetation, of rewetting (by raising the water table) areas planted with peatland plants.

Background

The surface of exploited peatlands can be too dry for natural peatland vegetation. Drainage for agriculture, peat extraction or construction – within or near to a focal peatland – can dry out the peat surface. To rewet surface peat, the water table of a large area of peatland could be raised by e.g. blocking drainage ditches, blocking underground channels or peat pipes, building raised banks, switching off drainage pumps or restoring inflows.

To be included as evidence in this section, studies must have experimentally tested the effect of raising the water table on planted peatland vegetation (e.g. by comparing rewetted areas to areas that remain drained).

Related interventions: rewet peatlands without introducing vegetation (Section 8.1); lower peatland surface by excavation/peat removal, bringing it closer to the water table (Sections 12.5 and 12.7); excavate/remove surface peat or irrigate peatland to complement planting (Sections 13.8–13.11); introduce peatland vegetation – overall effects (Sections 12.16–12.18).

13.8 Irrigate peatland (before/after planting)

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- **One study** examined the effect, on peatland vegetation, of irrigating areas planted with peatland plants. The study was in a bog.
- **Cover (1 study):** One replicated, paired, controlled, before-and-after study in a bog in Canada[1] found that irrigation increased the number of *Sphagnum* moss shoots present 1–2 growing seasons after sowing *Sphagnum* fragments.

Background

Irrigation systems, such as sprinklers, could be used to maintain a damp peat surface and stop planted vegetation from drying out (Rochefort & Bastien 1998). Water could be recirculated from drainage ditches or ponds on the peatland. Irrigation can be expensive so may be best used as a short-term intervention to kick-start restoration.

CAUTION: A suitable water source, with the right level of nutrients and acidity/alkalinity, must be chosen to avoid altering chemical conditions on the peatland (Lamers *et al.* 2002). For example, bogs should only be irrigated with water stored on the bog, not ground water. Taking water for irrigation might reduce water levels in neighbouring wetlands.

Related interventions: irrigate peatlands without introducing vegetation (Section 8.2); introduce peatland vegetation – overall effects (Sections 12.16–12.18).

Lamers L.P., Smolders A.J.P. & Roelofs J.G.M. (2002) The restoration of fens in the Netherlands. *Hydrobiologia*, 478, 107–130.

Rochefort L. & Bastien D.F. (1998) Réintroduction de sphaignes dans une tourbière exploitée: évaluation de divers moyens de protection contre la dessiccation (Reintroduction of *Sphagnum* to an exploited bog: evaluation of various methods for protection against desiccation; in French). *Écoscience*, 5, 117–127.

A replicated, paired, controlled, before-and-after study in 1993–1994 in a historically mined bog in Quebec, Canada (1) found that irrigating plots sown with *Sphagnum*-dominated vegetation fragments increased the number of *Sphagnum* moss shoots present. The effect was biggest after one growing season (irrigated: 250–630 shoots/m^2; not irrigated: 60–310 shoots/m^2) but persisted after two growing seasons (irrigated: 95–770 shoots/m^2; not irrigated: 50–390 shoots/m^2). Irrigation also increased the number of *Sphagnum* shoots in additional plots that were not sown (see

Section 8.2). In spring 1993, three pairs of plots were established on slightly drained, bare peat. Sections of each plot were sown with vegetation fragments, dominated by one of three *Sphagnum* moss species (250 fragments/m^2). Three plots (one plot/pair) were irrigated during the summer, using sprinklers and water stored on the bog. The other plots were not irrigated. In autumn 1993 and 1994, all *Sphagnum* shoots were counted in forty 30 x 30 cm quadrats/plot.

(1) Rochefort L. & Bastien D.F. (1998) Réintroduction de sphaignes dans une tourbière exploitée: évaluation de divers moyens de protection contre la dessiccation (Reintroduction of *Sphagnum* to an exploited bog: evaluation of various methods for protection against desiccation; in French). *Écoscience*, 5, 117–127.

13.9 Reprofile/relandscape peatland (before planting)

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- **Four studies** examined the effect, on peatland vegetation, of reprofiling or relandscaping before planting peatland plants. All four studies were in bogs.
- **Survival (1 study):** One replicated, paired, controlled study in a bog in Canada[1] found that survival of sown *Sphagnum* mosses was higher, after one growing season, in reprofiled basins than on raised plots.
- **Cover (3 studies):** Two replicated, controlled, before-and-after studies in bogs in Canada[2a,2b] found that reprofiled basins had higher *Sphagnum* cover than raised plots, 3–4 growing seasons after sowing *Sphagnum-dominated vegetation fragments*. However, one controlled study in a bog in Estonia[3] reported that total *Sphagnum* cover did not differ between reprofiled and raised plots, 1–2 years after sowing. All three studies[2a,2b,3] found that reprofiled and raised plots developed similar cover of other mosses/bryophytes and vascular plants.

Background

The peatland surface could be modified before introducing vegetation to create a more suitable environment for vegetation growth. In particular, local moisture levels could be raised by excavating shallow basins (removing <30 cm of surface peat and pushing it into ridges). At the same time, drier mounds of peat can be flattened. Steep gully sides or eroding slopes can be reprofiled into shallower, more stable slopes.

CAUTION: Heavy machinery used for landscaping may churn and compress the peat soil, damaging its structure. Removing surface peat from bogs may expose fen peat, which has different chemical properties to bog peat and will not (in the short term) support bog vegetation (Lindsay & Clough 2016).

Related interventions: reprofile/relandscape without introducing vegetation (Section 12.5); raise water table to complement planting (Section 13.7); create small mounds or hollows in peat before planting (Section 13.10); remove upper layer of peat/soil before planting, without further reprofiling (Section 13.11); introduce peatland vegetation – overall effects (Sections 12.16–12.18).

Lindsay R.A. & Clough J. (2016) *A Review of the Influence of Ombrotrophic Peat Depth on the Successful Restoration of Bog Habitat*. Scottish Natural Heritage Commissioned Report 925.

A replicated, paired, controlled, before-and-after study in 1995 in a historically mined raised bog in Quebec, Canada (1) found that reprofiling plots into depressions, before sowing *Sphagnum*-dominated vegetation fragments, increased the number of

Sphagnum moss shoots present. After one growing season, reprofiled plots contained more *Sphagnum* shoots (469–629/m^2) than raised plots (146/m^2). Amongst reprofiled plots, those with slopes covered in plastic sheeting contained more *Sphagnum* shoots (629/m^2) than those without sheeting (469/m^2). In May 1995, nine 8 m^2 plots (three blocks of three) were established on bare rewetted peat. Six plots (two plots/block) were situated at the base of excavated slopes (i.e. in the point of a 1 m deep "V"). Three plots (one plot/block) were on raised peat. Additionally, the reprofiled slopes above three plots were covered with plastic sheets. All plots were sown with vegetation fragments (mostly seven mixed *Sphagnum* species), freshly collected from the surface of nearby bogs, then mulched with straw. In October 1995, *Sphagnum* shoots were counted in 240 quadrats/plot, each 400 cm^2 and placed systematically.

A replicated, controlled, before-and-after study in 1996–1999 in a historically mined bog in Quebec, Canada (2a) found that reprofiling plots into basins, before sowing vegetation fragments, increased *Sphagnum* moss cover but not vascular plant cover. After four growing seasons, basins contained greater cover of all three focal *Sphagnum* species (56–62%) than raised plots (8–23%). Reprofiling had no effect on vascular plant cover (5% in basins and raised plots). In spring 1996, four 8 x 12 m plots were reprofiled into basins by pushing 20–25 cm of peat into ridges around each plot. Four plots were not reprofiled (remained raised). In May 1996, freshly collected vegetation fragments were sown onto all eight bare peat plots. Equally sized areas of each plot were sown with fragments dominated by rusty bog moss *Sphagnum fuscum*, Magellanic bog moss *Sphagnum magellanicum* or red bog moss *Sphagnum rubellum*. All plots were mulched with straw after sowing. In autumn 1999, vegetation cover was visually estimated in 72 quadrats, each 25 x 25 cm, across each plot.

A replicated, controlled, before-and-after study in 1996–1998 in a historically mined raised bog in Quebec, Canada (2b) found that reprofiling plots into basins, before sowing vegetation fragments, increased *Sphagnum* moss cover but had no effect on cover of other mosses or vascular plants. After three growing seasons, reprofiled plots had greater cover of both recorded *Sphagnum* species (41–52%) than raised plots (17–19%), but similar cover of other mosses (excavated: 2%; not excavated: 2%) and vascular plants (excavated: 4%; not excavated: 2%). In May 1996, freshly collected vegetation fragments were spread by hand onto 14 bare peat plots. Ten plots had been reprofiled into basins (4–20 m wide) bordered by peat ridges (30–60 cm high). Four 15 x 15 m plots were not reprofiled (remained raised). Vegetation fragments dominated by either rusty bog moss *Sphagnum fuscum* or red bog moss *Sphagnum rubellum* were sown in separate strips within each plot. All plots were mulched with straw after sowing. In autumn 1998, vegetation cover was visually estimated in 12–30 quadrats, each 25 x 25 cm, across each plot.

A controlled study in 2012–2014 in a historically mined bog in Estonia (3) found that reprofiling peat before sowing vegetation fragments did not significantly affect vegetation cover, with the exception of some *Sphagnum* moss species. After 1–2 years, plots with and without reprofiling had similar cover of vascular plants (23 vs 24%), total bryophytes (57 vs 50%) and *Sphagnum* moss (54 vs 47%; not statistically tested). However, the reprofiled plot had significantly greater cover of rusty bog moss *Sphagnum fuscum* (31 vs 21%) and significantly less cover of red bog moss *Sphagnum rubellum* (11 vs 17%). Sheathed cottongrass *Eriophorum vaginatum* and sedge *Carex* sp. were present at low cover (<1%) in both plots. In spring 2012, one of two adjacent bare peat plots was reprofiled (top 20 cm of peat pushed into ridges around the plot).

Both plots were rewetted (drainage ditch blocked), sown with vegetation fragments from a nearby bog and mulched with straw. In June and September 2013 and 2014, vegetation cover was estimated in ten 50 x 50 cm quadrats/plot.

(1) Bugnon J.-L., Rochefort L. & Price J.S. (1997) Field experiment of *Sphagnum* reintroduction on a dry abandoned peatland in eastern Canada. *Wetlands*, 17, 513–517.
(2) Campeau S., Rochefort L. & Price J.S. (2004) On the use of shallow basins to restore cutover peatlands: plant establishment. *Restoration Ecology*, 12, 471–482.
(3) Karofeld E., Müür M. & Vellak K. (2016) Factors affecting re-vegetation dynamics of experimentally restored extracted peatland in Estonia. *Environmental Science and Pollution Research*, 23, 13706–13717.

13.10 Create mounds or hollows (before planting)

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- **Three studies** examined the effect, on peatland vegetation, of creating peat mounds or hollows before planting peatland plants. Two studies were in bogs[1,3]. One was in a tropical peat swamp[2].
- **Growth (1 study):** One controlled study in a peat swamp in Thailand[2] reported that trees planted into mounds of peat grew thicker stems than trees planted at ground level.
- **Cover (2 studies):** Two replicated, randomized, paired, controlled, before-and-after studies in bogs in Canada[1,3] found that roughening the peat surface (by harrowing[1], ploughing[1], creating vehicle tracks[1] or adding peat blocks[3]) did not significantly affect cover of planted *Sphagnum* moss after 1–3 growing seasons.

Background

Vegetation may struggle to colonise flat bare peat. It can be damaged by extreme heat, sunlight, strong winds or prolonged flooding. In temperate peatlands, roughening the peat surface (e.g. by ploughing, creating vehicle tracks, or adding peat blocks) could create lower sheltered, moist and shaded habitats that are more suitable for plant growth. These processes can also break up any hard crust or compress loose, dry peat. Although raised areas might be less suitable for peatland vegetation initially, the idea is that peatland vegetation can spread from colonized depressions. In tropical peatlands, planting trees into mounds might improve their survival by reducing the duration of flooding and increasing oxygen supply to the roots (Wibisono *et al.* 2005). Mounds may naturally form in peat swamp forests from the roots of fallen trees.

CAUTION: Manipulating the peat surface may damage its physical structure. Repeated use of vehicles on soft, wet peat may be particularly damaging (see Chapter 7).

Related interventions: roughen peat surface without introducing vegetation (Section 12.6); large scale reprofiling/relandscaping before planting (Section 13.9); introduce peatland vegetation – overall effects (Sections 12.16–12.18).

Wibisono I.T.C., Siboro L. & Suryadiputra I.N.N. (2005) *Panduan Rehabilitasi dan Teknik Silvikultur di Lahan Gambut* (A Guide to Rehabilitation and Silvicultural Engineering on Peatlands; in Indonesian). Wetlands International Indonesia Programme & Wildlife Habitat Canada, Bogor.

A replicated, randomized, paired, controlled, before-and-after study in 1995–1996 in a historically mined raised bog in Quebec, Canada (1) found that roughening the peat surface, before sowing *Sphagnum*-dominated vegetation fragments, had no effect on *Sphagnum* moss cover. After 1–2 growing seasons, roughened and smooth plots had similar cover of *Sphagnum*, when compared amongst mulched areas (roughened: 1.4–4.7%; smooth: 1.2–2.3%) or unmulched areas (roughened: 0.1–0.3%; smooth: 0.1–0.2%). In May 1995, twelve 15 x 30 m plots were established, in three blocks of four, on bare rewetted peat. Three plots (one random plot/block) received each roughening treatment: harrowing (5 cm deep), ploughing (20 cm deep), using bulldozer tracks to create trenches (1 m wide, 20 cm deep), or no intervention (smooth plots). Then, all plots were sown with vegetation fragments (mostly *Sphagnum* moss) from the surface of a nearby bog. Half of each plot was mulched with straw. In June and September 1996, *Sphagnum* cover was estimated in 36–72 quadrats/plot, each 25 x 25 cm.

A controlled study in a degraded peat swamp in Thailand (2) reported that five tree species grew thicker stems when planted into mounds than when planted at ground level. The results were not tested for statistical significance. After three years and for all five planted species, trees planted into mounds had developed thicker stems (3–6 cm) than trees planted at ground level (2–3 cm). Mounds had a particularly strong effect on *Syzygium pyrifolium* stem thickness (mounded: 6 cm; ground level: 3 cm). In a degraded peat swamp, trees were either planted into mounds of peat (50 cm high, 70–90 cm circumference) or at ground level. After three years, the diameter of all trees was measured 10 cm above the peat surface. The year, number of trees and their initial size were not reported.

A replicated, randomized, paired, controlled, before-and-after study in 2007–2010 in two historically disturbed bogs in Ontario, Canada (3) found that placing peat blocks on a peatland, before sowing *Sphagnum* moss, had no effect on bryophyte cover. After three years, *Sphagnum* cover did not differ significantly between plots with peat blocks (40%) or without (38%). There was also no difference in total bryophyte cover between plots with peat blocks (66%) or without (67%). In August 2007, six pairs of 2 x 2 m plots were established on bare peat, historically disturbed by vehicles or pipeline construction. Twenty-five bare peat blocks (10 x 12 x 20 cm) were evenly spaced on one random plot in each pair. Then, all plots received fresh fragments of rusty bog moss *Sphagnum fuscum* and flat-topped bog moss *Sphagnum fallax*, and 30 g/m^2 rock phosphate fertilizer. In August 2010, moss cover was estimated in six random 12.5 x 12.5 cm subplots within each plot.

(1) Price J., Rochefort L. & Quinty F. (1998) Energy and moisture considerations on cutover peatlands: surface microtopography, mulch cover and *Sphagnum* regeneration. *Ecological Engineering*, 10, 293–312.
(2) Nuyim T. (2000) *Whole aspect on nature and management of peat swamp forest in Thailand*. Proceedings of the International Symposium on Tropical Peatlands, 22-23 November 1999, Bogor, Indonesia, 109–117.
(3) Corson A. & Campbell D. (2013) Testing protocols to restore disturbed *Sphagnum*-dominated peatlands in the Hudson Bay Lowland. *Wetlands*, 33, 291–299.

13.11 Remove upper layer of peat/soil (before planting)

B Ⓕ S

- We captured no evidence for the effect, on peatland vegetation, of removing the upper layer of peat or soil before planting peatland plants.

Background

The upper layer of soil/peat (and any vegetation on it) could be removed from damaged peatlands, creating a new surface of bare peat for introducing vegetation with fewer nutrients, no undesirable seed bank, and often wetter and less acidic peat since the surface is closer to the water table (Grootjans *et al.* 2002). Clearing surface peat could also remove a hard crust or loose peat through which water cannot easily rise to the plants above.

CAUTION: Soil stripping may be unsuitable for wetter peatlands as heavy machinery involved may churn and compress the peat soil. Stripping surface peat from bogs may expose fen peat, which has different chemical properties to bog peat and will not (in the short term) support bog vegetation (Lindsay & Clough 2016).

Related interventions: remove surface peat/soil without introducing vegetation (Section 12.7); reprofiling/relandscaping, e.g. building ridges or embankments, before planting (Section 13.9); bury surface peat/soil before planting (Section 13.12); introduce peatland vegetation – overall effects (Sections 12.16–12.18).

Grootjans A.P., Bakker J.P., Jansen A.J.M. & Kemmers R.H. (2002) Restoration of brook valley meadows in the Netherlands. *Hydrobiologia*, 478, 149–170.

Lindsay R.A. & Clough J. (2016) *A Review of the Influence of Ombrotrophic Peat Depth on the Successful Restoration of Bog Habitat*. Scottish Natural Heritage Commissioned Report 925.

13.12 Bury upper layer of peat/soil (before planting)

B F S

- We captured no evidence for the effect, on peatland vegetation, of burying the upper layer of peat or soil before planting peatland plants.

Background

The upper layer of soil or peat (and any vegetation on it) from damaged peatlands could be buried under deeper peat layers, for instance by deep ploughing. Burial can create bare peat with spaces for introduced plants to grow, prevent undesirable plants from growing from seeds already in the soil, and remove excess nutrients that favour growth of undesirable weedy plants (Glen *et al.* 2017). Inverting, rather than removing, the upper soil layer maintains the ground level.

CAUTION: Soil burial may be unsuitable for wetter peatlands as heavy machinery involved may churn and compress the peat soil. Burying surface peat from bogs may expose fen peat, which has different chemical properties to bog peat and will not (in the short term) support bog vegetation (Lindsay & Clough 2016).

Related interventions: bury surface peat/soil without introducing vegetation (Section 12.8); remove surface peat/soil before planting (Section 13.11); introduce peatland vegetation – overall effects (Sections 12.16–12.18).

Glen E., Price E.A.C., Caporn S.J.M., Carroll J.A., Jones L.M. & Scott R. (2017) Evaluation of topsoil inversion in UK habitat creation and restoration schemes. *Restoration Ecology*, 25, 72–81.

Lindsay R.A. & Clough J. (2016) *A Review of the Influence of Ombrotrophic Peat Depth on the Successful Restoration of Bog Habitat*. Scottish Natural Heritage Commissioned Report 925.

13.13 Add fresh peat to peatland (before planting)

B F S

- **One study** examined the effect, on peatland vegetation, of adding fresh peat before planting peatland plants. The study was in a bog.

- **Cover (1 study):** One replicated, controlled, before-and-after study in a bog New Zealand[1] reported that plots amended with fine peat supported higher cover of two sown plant species than the original (tilled) bog surface. However, for one species fertilization cancelled out this effect.

Background

Fresh ground or 'milled' peat could be added to the surface of peatlands before introducing vegetation. The fresh peat could provide a better substrate for plant growth: covering any dry crust on the degraded peatland and providing a fine substrate for rooting. Added peat can also be used to smooth over the peatland surface (removing raised areas or depressions that might be too dry or wet for peatland plant growth), modify the water level to which introduced vegetation is exposed (e.g. to avoid winter flooding), or create a buffer between polluted peat and the surface where plants are introduced (Rezanezhad *et al.* 2012). Finally, the fresh peat may contain extra nutrients to give plants an initial boost.

Related interventions: introduce seeds of peatland vegetation, including addition of smaller amounts of soil to introduce the seeds/spores within (Section 12.18).

Rezanezhad F., Andersen R., Pouliot R., Price J.S., Rochefort L. & Graf M.D. (2012) How fen vegetation structure affects the transport of oil sands process-affected waters. *Wetlands*, 32, 557–570.

A replicated, controlled, before-and-after study in 1998–2000 in a historically mined raised bog in New Zealand (1) reported that amending plots with fine peat allowed greater cover of two sown species to develop, although for one species only in the absence of fertilizer. These results are not based on tests of statistical significance. After 810 days, plots amended with peat before sowing manuka *Leptospermum scoparium* seeds had 99–100% manuka cover, compared to only 1–68% manuka cover in plots not amended with peat. Plots amended with peat before sowing bamboo rush *Sporadanthus ferrugineus* seeds developed 0–32% rush cover (0–6% when fertilized; 32% when not fertilized), compared to 0–11% when not amended with peat. In March 1998, forty-eight 25 m^2 plots were sown: 24 with manuka and 24 with bamboo rush. For each plant species, eight plots were on a 30 cm layer of fresh fine peat and 16 directly on the existing bare peat (but note this was also tilled). Some plots were fertilized with phosphorous, nitrogen or both. In June 2000, canopy cover was visually estimated in each plot. This study also reported the effect of fertilization in unsown plots (see original paper).

(1) Schipper L.A., Clarkson B.R., Vojvodic-Cukovic M. & Webster R. (2002) Restoring cut-over restiad peat bogs: a factorial experiment of nutrients, seed and cultivation. *Ecological Engineering*, 19, 29–40.

13.14 Encapsulate planted moss fragments in beads/gel Ⓑ Ⓕ S

- We captured no evidence for the effect of encapsulating moss fragments on their performance, relative to loose moss fragments, when introduced to peatlands.

Background

Sphagnum fragments can be encapsulated in gel beads or a suspended in a gel slurry (e.g. www.beadamoss.co.uk). The gel keeps the moss fragments moist and provides an initial food source. It may also make sowing easier and cheaper. After spreading, the gel eventually breaks down. Qualitative observations indicate that encapsulated fragments may survive for longer than loose fragments (Hinde *et al.* 2010). One study in the UK recorded fewer *Sphagnum* clumps in plots sown with beads than with loose fragments, but this compared different species sown at different densities (Rosenburgh 2015). We captured no direct quantitative comparisons of encapsulated and loose moss performance.

Related interventions: spread peatland vegetation onto peat surface (Section 12.17).

Hinde S., Rosenburgh A., Wright N., Buckler M. & Caporn S. (2010) Sphagnum *Re-introduction Project: a Report on Research into the Re-introduction of* Sphagnum *Mosses to a Degraded Moorland*. Moors for the Future Research Report 18.

Rosenburgh A. (2015) *Restoration and recovery of* Sphagnum *on degraded blanket bog*. PhD Thesis. Manchester Metropolitan University.

13.15 Use fences or barriers to protect planted vegetation Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect of using fences or barriers to protect planted peatland vegetation.

Background

Small plants introduced to peatlands may be vulnerable to grazing by livestock or wild herbivores. Barriers could be used to exclude these animals. We consider all kinds of barriers in this section, from those erected around plants (e.g. fences and tree guards) to those placed on plants (e.g. copper strips to deter slugs and snails, and sticky/oily substances that may be applied to tree trunks). Barriers erected around individual plants (e.g. tree guards) may also offer some protection against desiccation, sunlight, temperature fluctuations and strong winds.

Related interventions: use fences or barriers to protect peatland vegetation in general, by excluding livestock (Sections 3.5 and 3.6) or wild herbivores (Section 9.12).

13.16 Remove vegetation that could compete with planted peatland vegetation Ⓑ Ⓕ Ⓢ

- **One study** examined the effect of removing competing plants to aid planted peatland vegetation. The study was in a bog.
- **Survival (1 study):** One controlled study in a bog in the UK[1] reported that some *Sphagnum* moss survived when sown (in gel beads) into a plot where purple moor grass had previously been cut, but no moss survived in a plot where grass had not been cut.

Background

Removing other plants before or after planting peatland vegetation could reduce competition for space, light and nutrients. Survival and growth of planted vegetation may be improved. Note that abundant competitors, and/or the absence of the vegetation to be introduced, could be symptoms of inappropriate physical conditions that may also need to be managed.

CAUTION: Existing vegetation may help planted vegetation to establish, protecting it from extreme temperatures, strong sunlight and desiccation.

Related interventions: interventions to control competing plants without introducing vegetation (Chapters 8 and 9); introduce nurse plants (Sections 12.14 and 12.6).

A controlled study in 2010–2013 in a degraded, grassy blanket bog in England, UK (1) reported that some sown *Sphagnum* moss survived in a plot where purple moor grass *Molinia caerulea* had previously been cut, but no moss survived in an uncut plot. This result was not tested for statistical significance. After three years, a plot that was flailed before sowing *Sphagnum* contained 28 *Sphagnum* clumps (0.03% cover). No *Sphagnum* survived in an adjacent plot that was not flailed before sowing. In October 2010, two adjacent 3 x 3 m plots were sown with flat-topped bog moss *Sphagnum fallax*, encapsulated in gel beads (400 beads/m^2). Both plots were dominated by purple moor grass, but one was flailed (cut) before sowing. Grass litter was left in place. In September 2013, *Sphagnum* clumps were identified in each plot and their area was measured.

(1) Rosenburgh A. (2015) *Restoration and recovery of* Sphagnum *on degraded blanket bog*. PhD Thesis. Manchester Metropolitan University.

13.17 Add root-associated fungi to plants (before planting) Ⓑ Ⓕ Ⓢ

- **Three studies** examined the effect of adding root-associated fungi to planted peatland vegetation. All three studies involved peat swamp tree seedlings: two in the wild[1,2] and one in a nursery[3].
- **Survival (2 studies):** Two controlled studies (one also replicated, paired, before-and-after) in peat swamps in Indonesia[1,2] found that adding root fungi did not affect survival of planted red balau[1,2] or jelutong[2] in all or most cases. However, one fungal treatment increased red balau survival in one study[1].
- **Growth (3 studies):** Two replicated, controlled, before-and-after studies (one also paired) of peat swamp trees in Indonesia found that adding root fungi to seedlings had no effect on growth: for red balau and jelutong[2] or the majority of 15 tested species[3]. However, one controlled study in Indonesia[1] found that adding root fungi increased growth of red balau seedlings.

Background

Many plants (including grasses, trees and shrubs) form mutually beneficial associations with fungi. The 'mycorrhizal' fungi live in or around plant roots. They can increase plant access to nutrients and minimise the effect of stresses such as drought and pollution (Finlay 2008). Adding these fungi to plants before they are introduced to peatlands could therefore help survival and growth. Fungi could be added via a root dip, or through adding spores to soil in the nursery.

Related interventions: other interventions to protect or prepare plants (Section 13.18); introduce peatland vegetation – overall effects (Sections 12.16–12.18).

Finlay R.D. (2008) Ecological aspects of mycorrhizal symbiosis: with special emphasis on the functional diversity of interactions involving the extraradical mycelium. *Journal of Experimental Botany*, 59, 1115–1126.

A controlled study in 2002–2006 in a logged peat swamp in Kalimantan, Indonesia (1) found that inoculating red balau *Shorea balangeran* seedlings with root fungi increased growth (for three of three fungal species) but did not affect survival (for two of three fungal species). Forty months after planting, inoculated seedlings were taller than uninoculated seedlings (213–240 cm vs 206 cm) and had wider stems (diameter 30–37 cm vs 27 cm). Only seedlings inoculated with *Strobilomyces* sp. fungi had higher survival (85%) than uninoculated seedlings (83%). Survival of seedlings inoculated with two other fugal species was 79–81%. In November 2002, 400 red balau seedlings were planted into logged forest: 100 inoculated with each fungal species and 100 uninoculated. Seedlings had been grown in sterilized peat in a nursery and inoculated with wild-collected spores suspended in water. Seedling height, stem diameter and survival were measured 40 months after planting.

A replicated, paired, controlled, before-and-after study in 2007–2009 in a peat swamp forest in Indonesia (2) found that inoculation with root fungi had no effect on survival or growth of two planted tree species: red balau *Shorea balangeran* and jelutong *Dyera polyphylla*. One year after planting, seedlings with and without added root fungi had similar survival (75–91%; data not reported separately for forest types), similar height increase (in five of five forest types; with fungi: 2–11 cm; without fungi: 3–10 cm) and similar diameter increase (in five of five forest types; with fungi: 0.6–2.7 mm; without fungi: 0.7–2.4 mm). In 2007 or 2008, nursery-reared seedlings (800 red balau and 700 jelutong) were planted in five forest types from natural/closed forest to degraded/open land. Approximately two thirds of these seedlings had been inoculated with fungi by adding spore tablets to the soil in the nursery. The other seedlings were not inoculated. After one year, seedling survival and growth were measured.

A replicated, controlled, before-and-after study in 2011 in a nursery in Indonesia (3) found that inoculation with root fungi typically had no effect on growth of peat swamp tree seedlings. Seedlings of 15 species were studied. Seedlings with and without added root fungi showed similar height growth for 14–15 species (depending on the fungus used) and similar stem diameter growth for 11–14 species. In June 2011, thirty seedlings of each tree species were inoculated with root fungi (10 seedlings for each of three fungal species). Ten additional seedlings were not inoculated. Seedlings were planted in pots of sterilized peat, having been grown from sterilized seed or transplanted from the wild. The duration of the experiment was not reported.

(1) Turjaman M., Santoso E., Susanto A., Gaman S., Limin S.H., Tamai Y., Osaki M. & Tawaraya K. (2011) Ectomycorrhizal fungi promote growth of *Shorea balangeran* in degraded peat swamp forests. *Wetlands Ecology and Management*, 19, 331–339.
(2) Graham L.L.B., Turjaman M. & Page S.E. (2013) *Shorea balangeran* and *Dyera polyphylla* (syn. *Dyera lowii*) as tropical peat swamp forest restoration transplant species: effects of mycorrhizae and level of disturbance. *Wetlands Ecology and Management*, 21, 307–321.
(3) Yuwati T.W., Graham L.L.B., Rachmanadi D., Santosa P.B. & Rusmana (2014) Response of peat swamp forest species to mycorrhizal inoculations. Pages 64–76 in: Banjarbaru Forestry Research Unit, FORDA & L.L.B. Graham (eds.) *Tropical Peat Swamp Forest Silviculture in Central Kalimantan*. Kalimantan Forests and Climate Partnership, Indonesia.

13.18 Protect or prepare vegetation before planting (other interventions) Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect of protecting or preparing peatland vegetation before planting (other than by adding root-associated fungi).

Background

The handling of plants before planting can affect their germination, survival or growth rates. This section considers the effects of treatments applied to seedlings, seeds or other planted material before planting. For example, plant roots might be dipped in a substance to keep them moist during transportation to a site. Plants might be treated with hormones before planting to stimulate their growth. Seed germination rates might be improved by stratification (exposing seeds to a period of chilling) or scarification (damaging the seed coating to allow water to enter the seed).

Related interventions: add root-associated fungi to plants before planting (Section 13.17); introduce peatland vegetation – overall effects (Sections 12.16–12.18).

14. Habitat protection

Background

This chapter considers protection of peatlands through legislation, voluntary agreements and economic incentives. Legal protection could be designated locally (e.g. state conservation policies in Canada; Poulin *et al.* 2004), nationally (e.g. Chilean conservation policies that sometimes cover peatlands; Möller & Muñoz-Pedreros 2014) or internationally (e.g. European Union Natura 2000 network of protected areas). Note that protection on paper does always confer protection in reality.

These protection mechanisms are designed to prevent damage, loss or fragmentation of peatlands from multiple threats, such as those in Chapters 2–10. This kind of intervention *might be the only intervention necessary* for pristine or relatively undisturbed peatlands. Active management of such pristine peatlands can often cause more harm than good. (Broad-scale interventions to reduce threats, such as increasing use of renewable energy sources to reduce greenhouse gas emissions, could also help conserve pristine peatlands but are beyond the scope of this synopsis).

Related interventions: physical protection of peatlands e.g. by building fences, walls or barriers (Chapter 3, Chapter 10 and Chapter 11); education and awareness interventions designed to protect peatlands (Chapter 15).

Möller P. & Muñoz-Pedreros A. (2014) Legal protection assessment of different inland wetlands in Chile. *Revista Chilena de Historia Natural*, 87, 23.

Poulin M., Rochefort L., Pellerin S. & Thibault J. (2004) Threats and protection for peatlands in Eastern Canada. *Géocarrefour*, 79, 331–344.

Key messages

14.1 *Legally protect peatlands* **5 studies**

Peatland habitat: Two studies in Indonesia reported that peat swamp forest was lost from within the boundaries of national parks. However, one of these studies reported that forest loss was greater outside the national park. One before-and-after study in China reported that peatland area initially decreased following legal protection, but increased in the longer term.

Plant community composition: One before-and-after study in a bog in Denmark reported that the plant community composition changed over 161 years of protection. In particular, woody plants became more abundant.

Vegetation cover: One site comparison study in Chile found that protected peatland had greater vegetation cover (total, herbs and shrubs) than adjacent grazed and moss-harvested peatland.

Overall plant richness/diversity: One before-and-after study in Denmark reported that the number of plant species in a protected bog fluctuated over time, with no clear trend. One site comparison study in Chile found that protected peatland had lower plant richness and diversity, but also fewer non-native species, than adjacent grazed and harvested peatland.

14.2 *Create legislation for 'no net loss' of wetlands* **0 studies**

We captured no evidence for the effect on peatland habitats of creating legislation for no net loss of wetlands.

14.3 ***Adopt voluntary agreements to protect peatlands***	**0 studies**

We captured no evidence for the effect on peatland habitats of adopting voluntary agreements to protect them.

14.4 ***Pay landowners to protect peatlands***	**1 study**

Peatland habitat: One review reported that agri-environment schemes in the UK had mixed effects on bogs, protecting the area of bog habitat in three of six cases.

14.5 ***Increase 'on-the-ground' protection (e.g. rangers)***	**1 study**

Behaviour change: One before-and-after study in a peat swamp forest in Indonesia reported that the number of illegal sawmills decreased over two years of anti-logging patrols.

14.6 ***Allow sustainable use of peatlands***	**0 studies**

We captured no evidence for the effect on peatland habitats of allowing sustainable use.

Interventions

14.1 Legally protect peatlands

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- **Five studies** examined the effect on peatland habitats of legally protecting them: two of tropical peat swamp forest[1,4], two of unspecified peatlands[2,5] and one of a bog[3].
- **Peatland habitat (3 studies):** Two studies in Indonesia[1,4] reported that peat swamp forest was lost from within the boundaries of national parks. However, one of these studies[4] was a site comparison and reported that forest loss was greater outside the national park. One before-and-after study of peatlands in China[2] reported that peatland area initially decreased, but then increased, following legal protection.
- **Plant community composition (1 study):** One before-and-after study in a bog in Denmark[3] reported that the plant community composition changed over 161 years of protection. In particular, woody plants became more abundant.
- **Vegetation cover (1 study):** One site comparison study in a peatland in Chile[5] found that a protected area had greater vegetation cover (total, herbs and shrubs) than an adjacent grazed and moss-harvested area.
- **Overall plant richness/diversity (2 studies):** One before-and-after study in Denmark[3] reported that the number of plant species in a protected bog fluctuated over time, with no clear trend. One site comparison study in a peatland in Chile[5] found that a protected area had lower plant richness and diversity (but also fewer non-native species) than an adjacent grazed and harvested area.

Background

Peatland habitats might be protected by law. Specific sites may be purchased or designated as protected areas in order to limit damaging activities like trampling, vehicle use, extraction of resources and development. Protected sites may be left alone or, where necessary, actively managed/restored to increase their value. This section considers the overall effects of legally protecting specific peatland areas. Effects of individual interventions performed within protected areas are also considered under the relevant section.

Assessing the effectiveness of protected areas is particularly difficult. For example, protected and unprotected areas might start off with different quality habitats (protection being granted to the best quality peatlands). Protected areas are also more likely to be in remote areas, so less accessible to threats such as harvesting (Joppa & Pfaff 2009). Finally, effectiveness is best monitored over long timescales, but this increases the chance that other factors influence the ecosystem. The most reliable studies would compare protected and unprotected areas over time, and possibly correct for some of the biases.

Related interventions: create legislation for 'no net loss' of wetlands (Section 14.2); adopt voluntary codes to protect peatlands (Section 14.3).

Joppa L.N. & Pfaff A. (2009) High and far: biases in the location of protected areas. *PLoS ONE*, 4, e8273.

A study in 1990–2004 of a national park in Indonesia, including peat swamp and lowland forest (1) reported that legal protection did not prevent deforestation. Gunung Palung National Park was designated in 1990. The rate of forest loss within the national park increased from 1,200 ha/year in 1994 to 9,000 ha/year in 2002. These estimates include both lowland forest (on non-peat soils) and peat swamp forest, but deforestation did occur in both forest types (data presented as maps). Forest cover in the national park was mapped using satellite images (30 m resolution) taken between 1994 and 2002. Land cover classification was validated using finer resolution satellite images, aerial photographs and field surveys.

A before-and-after study in 1990–2009 of peatlands on the Zoige Plateau, China (2) reported that following legal protection of the Plateau, the area of peatland vegetation decreased initially but increased in the longer term. In 1990, the study area contained 4,143 km^2 of peatland vegetation. The Plateau was designated as a Natural Reserve in 1994 then upgraded to a National Nature Reserve in 1998. In 2000, the area of peatland vegetation had shrunk by 18% to 3,407 km^2. However, by 2009 this had increased by 5% to 3,589 km^2. The area of peatland vegetation was calculated from satellite images. The study noted that grazing land has been abandoned since the Plateau was protected, although it is not clear to what extent this is directly related to the protection. The study also noted changes in temperature and rainfall over time.

A before-and-after study in 1844–2005 in a historically mined raised bog in Denmark (3) reported that following legal protection, the plant community changed over time in favour of woody species, whilst plants species richness fluctuated without trend. These results were not tested for statistical significance. Over 161 years of protection, the overall composition of the plant community changed (data reported as a graphical analysis). In particular, tree/shrub abundance increased (overall, and for 17 of 20 species). In 2005, the most common trees were downy birch *Betula pubescens* and common oak *Quercus robur* (both in 100% of monitored plots). The most common moss was *Sphagnum fallax* (in 14% of monitored cells). The number of vascular plant species in the bog fluctuated over time, with no clear trend (40 species before protection; 75 species after 41 years; 18 species after 127 years; 38 species after 161 years). In 1844, a mined bog was legally protected from further human use. Between 1844 and 2005, plant species were recorded in 18 permanent 113 m^2 plots. In 2005, moss presence was recorded in six 0.25 m^2 quadrats.

A site comparison study in 1973–2009 in peat swamp forest in Indonesia (4) reported that a legally protected area retained more forest cover than an adjacent unprotected area. The results were not tested for statistical significance. In the 1970s, 99% of Berbak National Park was covered by peat swamp forest (and 95% by nearly pristine forest). By 2009, total peat swamp forest cover had declined to 77% (and nearly pristine cover to 73%). However, these declines were smaller than outside the National Park (total: from 91 to 46%; nearly pristine: from 86 to 25%). In 2009, there were also fewer industrial plantations and smallholder farms inside the National Park (0% cover) than outside (21% cover). Land cover was mapped using satellite images (10–60 m resolution) taken between 1973 and 2009. The images covered 1,262 km^2 of Berbak National Park (protected as a game reserve since 1935 and a Ramsar site since 1992) and 2,128 km^2 of adjacent land.

A site comparison study in 2014 in a peatland in Chile (5) found that a protected area had greater vegetation cover and taller vegetation, but lower vascular plant richness and diversity, than an adjacent grazed and harvested area. The protected area had greater cover than the unprotected area of total vegetation (87 vs 62%), herbs (68 vs 51%) and shrubs (19 vs 11%) and contained taller vegetation (65 vs 13 cm). The protected area had lower vascular plant species richness than the unprotected area (7 vs 11 species/4 m^2) and lower diversity (reported as a diversity index), but also contained fewer non-native species (<0.1 vs 1.9 species/4 m^2). In 2014, vegetation cover and height were recorded in forty-four 2 x 2 m quadrats. Fifteen quadrats were in a protected part of a peatland (5.5 ha owned by a research station, fenced to exclude livestock for eight years and with no moss harvesting for at least 20 years). Twenty-nine quadrats were in an unprotected part (10.5 ha, grazed by four oxen and harvested every month).

(1) Curran L.M., Trigg S.N., McDonald A.K., Astiani D., Hardiono Y.M., Siregar P., Caniago I. & Kasischke E. (2004) Lowland forest loss in protected areas of Indonesian Borneo. *Science*, 303, 1000–1003.
(2) Yao L., Zhao Y., Gao S., Sun J. & Li F. (2011) The peatland area change in past 20 years in the Zoige Basin, eastern Tibetan Plateau. *Frontiers in Earth Science*, 5, 271–275.
(3) Kollmann J. & Rasmussen K.K. (2012) Succession of a degraded bog in NE Denmark over 164 years – monitoring one of the earliest restoration experiments. *Tuexenia*, 32, 67–85.
(4) Miettinen J., Wang J., Hooijer A. & Liew S. (2013) Peatland conversion and degradation processes in insular Southeast Asia: a case study in Jambi, Indonesia. *Land Degradation and Development*, 24, 334–341.
(5) Cabezas J., Galleguillos M., Valdés A., Fuentes J.P., Pérez C. & Perez-Quezada J.F. (2015) Evaluation of impacts of management in an anthropogenic peatland using field and remote sensing data. *Ecosphere*, 6, 1–24.

14.2 Create legislation for 'no net loss' of wetlands

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- We captured no evidence for the effect on peatland habitats of creating legislation for no net loss of wetlands.

Background

Legislation may be developed to ensure no net loss of habitat. Simply, this means that any habitat lost to development must be compensated for by restoration or creation elsewhere. For example, under the Clean Water Act in the USA, a permit for development on wetlands may only be granted if there are plans to restore or create other wetlands. Alternatively, wetlands can be 'bought' from a 'bank' of restoration or creation projects to offset damage done by a particular development (Burgin 2010).

Related interventions: legally protect peatlands (Section 14.1); habitat creation and restoration (Chapter 12).

Burgin S. (2010) 'Mitigation banks' for wetland conservation: a major success or an unmitigated disaster. *Wetlands Ecology and Management*, 18, 49–55.

14.3 Adopt voluntary agreements to protect peatlands

Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect on peatland habitats of adopting voluntary agreements to protect them.

Background

Voluntary agreements or codes (i.e. not legally binding) could be used to protect peatlands. Such agreements prevent or restrict activities that damage specific peatlands. They may apply within a focal peatland or in surrounding habitats to prevent threats from spilling over into peatlands. Additionally, voluntary agreements often encourage peatland restoration or creation to protect the overall peatland resource. For example, members of the Roundtable on Sustainable Palm Oil (RSPO) are prohibited from clearing primary forest, discouraged from using fire to clear land, encouraged to leave corridors to connect forest patches and encouraged to restore or rehabilitate neighbouring peatland (Parish *et al.* 2012). The Ramsar Convention provides the basis for voluntary agreements to protect wetlands (including peatlands), although it is often translated into legal protection by national legislation (Ramsar Convention Secretariat 2007).

This section considers the overall effects of protecting peatlands with voluntary agreements (but excluding information on uptake only e.g. area of land or number of people signed up to agreements). Effects of individual interventions performed under voluntary agreements are considered elsewhere.

Parish F., Lim S.S., Perumal B. & Giesen W. (eds.) (2012) *RSPO Manual on Best Management Practices (BMPs) for Management and Rehabilitation of Natural Vegetation Associated with Oil Palm Cultivation on Peat*. Roundtable on Sustainable Palm Oil, Kuala Lumpur.

Ramsar Convention Secretariat (2007) *How States May Join the Ramsar Convention*. Ramsar Information Paper No. 13.

14.4 Pay landowners to protect peatlands

Ⓑ Ⓕ Ⓢ

- **One study** examined the effect on peatland habitats of paying landowners to protect them. The study was of bogs.
- **Peatland habitat (1 study):** One review reported that agri-environment schemes in the UK[1] had mixed effects on bogs, protecting the area of bog habitat in three of six cases.

Background

Landowners could be paid to protect natural, vegetated peatlands and the benefits they provide (e.g. carbon storage). Landowners may be paid to protect pristine peatlands. Where necessary, they may be paid to actively manage areas to maintain or restore peatlands. Thus, many payment schemes also protect the peatland resource as a whole. Payments could be made directly or as tax incentives, could be paid as cash or as alternative lands, and could come from governments, non-governmental organizations or private sponsorship. Payments should be supported by advice, monitoring and enforcement.

Examples of payment schemes relevant to peatlands include the Bio-Rights programme (van Eijk & Kumar 2009), the UK Peatland Code and the German

MoorFutures® voluntary carbon market (Bonn et al 2014) and nationally or internationally funded agri-environment schemes (Keenleyside & Moxey 2011).

This section considers the overall effects of payment schemes on peatland habitats (but excluding information on uptake only e.g. area of land managed under payment schemes, or number of people signed up). Effects of individual interventions performed under payment schemes are considered elsewhere.

Related intervention: adopt ecotourism principles/create an ecotourism site as a source of funding to protect natural peatlands (Section 7.7).

Bonn A., Reed M.S., Evans C.D., Joosten H., Bain C., Farmer J., Emmer I., Couwenberg J., Moxey A., Artz R., Tanneberger F., von Unger M., Smyth M.-A. & Birnie D. (2014) Investing in nature: developing ecosystem service markets for peatland restoration. *Ecosystem Services*, 9, 54–65.

van Eijk P. & Kumar R. (2009) *Bio-Rights in Theory and Practice. A Financing Mechanism for Linking Poverty Alleviation and Environmental Conservation*. Wetlands International, Wageningen.

Keenleyside C. & Moxey A. (2011) *Public Funding of Peatland Management and Restoration in the UK – a Review*. Report to IUCN UK Peatland Programme, Edinburgh.

A 2008 review of agri-environment schemes in Scotland, UK (1) reported mixed effects on the area of bog habitat. No statistical tests were carried out. In three of six areas, agri-environment schemes protected bog habitats. The area of bog increased more, or decreased less, than would be expected based on national trends on similar land. In the other three areas, agri-environment schemes did not protect bog habitats. Here, the area of bog decreased more, or increased less, than would be expected based on national trends in similar land. The review does not report details of the agri-environment schemes.

(1) Boatman N., Ramwell C., Parry H., Jones N., Bishop J., Gaskell P., Short C., Mills J. & Dwyer J. (2008) *A Review of Environmental Benefits Supplied by Agri-Environment Schemes.* Land Use Policy Group Report FST20/79/041.

14.5 Increase 'on-the-ground' protection (e.g. rangers) Ⓑ Ⓕ Ⓢ

- **One study** examined the effect on peatland habitats of increasing 'on-the-ground' protection. The study was in tropical peat swamps.
- **Behaviour change (1 study):** One before-and-after study in a peat swamp forest in Indonesia[1] reported that the number of illegal sawmills decreased over two years of anti-logging patrols.

Background

This section considers the use of an 'on-the-ground' human presence to protect peatlands from immediate threats. This includes rangers or wardens that may patrol peatlands, ensuring legislation and voluntary agreements are followed. It also includes creating teams to directly manage threats e.g. firefighters.

Related interventions: raise awareness through engaging volunteers in peatland management (Section 15.2); adopt ecotourism principles/create an ecotourism site (Section 7.7).

A before-and-after study in 2006–2008 in peat swamp forest in Indonesia (1) reported that anti-logging patrols reduced the number of sawmills (associated with illegal logging) in the study area. Before patrols began, 147 sawmills were identified in Katingan Regency, including 102 in active use. After two years of weekly patrols, there were only two sawmills in the area. Patrols involved representatives of the National Park authorities, local communities and non-governmental organizations.

(1) CKPP (2008) *Provisional Report of the Central Kalimantan Peatland Project*. November 2008.

14.6 Allow sustainable use of peatlands

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- We captured no evidence for the effect on peatland habitats of allowing sustainable use.

Background

'Wise use' of wetlands is one of the fundamental principles of the Ramsar Convention (Ramsar Convention Secretariat 2010). Allowing sustainable use of peatlands could give them an economic value, preventing their conversion to other land uses (e.g. agriculture, mining or urban development). Meanwhile sustainable use does not, by definition, damage the peatland: use can be sustained year after year. Reeds could be harvested sustainably from fens, *Sphagnum* harvested from bogs and non-timber forest products (e.g. resin, latex, dyes, medicinal plants) harvested from tropical peat swamps. Sustainable use could complement other interventions e.g. pools created by blocking drainage canals could be used for fish farming (Suryadiputra *et al.* 2005).

This section considers the overall effects of allowing sustainable use of peatlands e.g. by granting licenses for long-term harvesting or requiring limited extraction of resources. In Australia, long-term *Sphagnum* harvesting licenses have been granted to encourage sustainable use (Whinam & Buxton 1997). In Indonesia, natural forest timber concessions are leased out on a long-term basis. Managers are obliged to maintain natural forest cover. Across all forest types (including some peat swamps), these concessions are as effective as strictly protected areas in preventing forest loss (Gaveau *et al.* 2013). Effects of individual interventions performed within sustainably managed peatlands are considered elsewhere.

Related interventions: mosaic management of agriculture or harvesting wild resources (Sections 3.1 and 6.5); provide new technologies to reduce pressure on wild resources (Section 6.6); adopt ecotourism principles or create an ecotourism site, as another way to add economic value to natural peatlands (Section 7.7); provide education or training programmes about sustainable management (Section 15.3).

Croon F.W. (2013) Saving reed lands by giving economic value to reed. *Mires and Peat*, 13, Article 10.

Gaveau D.L.A., Kshatriya M., Sheil D., Sloan S., Molidena E., Wijaya A., Wich S., Ancrenaz M., Hansen M., Broich M., Guariguata M.R., Pacheco P., Potapov P., Tubanova S. & Meijaard E. (2013) Reconciling forest conservation and logging in Indonesian Borneo. *PLoS ONE*, 8, e69887.

Ramsar Convention Secretariat (2010) *Wise Use of Wetlands: Concepts and Approaches for the Wise Use of Wetlands, Fourth Edition*. Ramsar Convention Secretariat, Gland, Switzerland.

Suryadiputra I.N.N., Alue Dohong R., Waspodo S.B., Muslihat L., Lubis I.R., Hasudungan F. & Wibisono I.T.C. (2005) *A Guide to the Blocking of Canals and Ditches in Conjunction with the Community*. Wetlands International Indonesia & Wildlife Habitat Canada, Bogor.

Whinam J. & Buxton R.P. (1997) *Sphagnum* peatlands of Australasia: an assessment of harvesting sustainability. *Biological Conservation,* 82, 21-29.

15. Education and awareness

Background

Education and awareness-raising programmes can teach people about the value of peatlands and suitable techniques for their management. The Ramsar Convention, for example, recognises the importance of Communication, Capacity Building, Education, Participation and Awareness with its dedicated CEPA programme (Ramsar Convention Secretariat 2017). Awareness-raising activities may be timed to coincide with World Wetlands Day: February 2nd every year.

Ideally, the effectiveness of such programmes would be measured quantitatively as an *impact* on peatland habitats (Thomson & Hoffman n.d.). More often, a change in behaviour, knowledge, awareness or attitude is measured. Studies that measure any of these *outcomes* have been included in this chapter. We assume they would ultimately translate into a benefit for peatland habitats. In contrast, studies that simply report *outputs* (e.g. the number of leaflets produced, or the number of people involved in an education programme) are not included in this chapter. We consider these measures too far removed from any actual impact on peatland vegetation.

Related interventions: legal protection of peatlands, which could offer opportunities for public education and increased awareness (Chapter 14). Volunteers could be engaged to carry out interventions in many of the other chapters, providing an opportunity for education and awareness-raising.

Ramsar Convention Secretariat (2017) The Ramsar CEPA Programme. Available at http://www.ramsar.org/activity/the-ramsar-cepa-programme. Accessed 10 October 2017.

Thomson G. & Hoffman J. (n.d.) *Measuring the Success of Environmental Education Programs*. Canadian Parks and Wilderness Society and Sierra Club of Canada, Ottawa.

Key messages

15.1.1 *Raise awareness amongst the public (general)* **1 study**

Behaviour change: One before-and-after study in the UK reported that following awareness-raising activities (e.g. publishing reports, organizing seminars and using education volunteers on garden centres), the percentage of the public buying peat-free compost increased.

15.1.2 *Raise awareness amongst the public (wild fire)* **0 studies**

We captured no evidence for the effect of interventions to raise awareness about wild fire on knowledge, behaviour, peatland habitats or peatland vegetation.

15.1.3 *Raise awareness amongst the public (problematic species)* **0 studies**

We captured no evidence for the effect of interventions to raise awareness about problematic species on knowledge, behaviour, peatland habitats or peatland vegetation.

15.2 *Raise awareness through engaging volunteers in management or monitoring* **0 studies**

We captured no evidence for the effect of engaging volunteers to manage or monitor peatlands on knowledge, behaviour, peatland habitats or peatland vegetation.

15.3 *Provide education or training about peatlands or peatland management* **2 studies**

Behaviour change: One study in peat swamps in Indonesia reported that over 3,500 households adopted sustainable farming practices following workshops about sustainable farming. One before-and-after study, also in peat swamps in Indonesia, reported that a training course on rubber farming increased the quality of rubber produced by local farmers.

15.4 *Lobby, campaign or demonstrate to protect peatlands* **2 studies**

Peatland protection: Two studies in the UK reported that the area of protected peatland increased following pressure from a campaign group (including business meetings, parliamentary debates, publishing reports and public engagement).

Behaviour change: One study in the UK reported that following pressure from the same campaign group, major retailers stopped buying compost containing peat from important peatland areas and horticultural companies began marketing peat-free compost.

Attitudes/awareness: One study in the UK reported that following pressure from the same campaign group, garden centres and local governments signed voluntary peatland conservation agreements.

Interventions

15.1 Raise awareness about peatlands amongst the public Ⓑ Ⓕ Ⓢ

Background

The public could be educated about the importance of peatlands, threats they face and what can be done to protect them. Messages could be conveyed through information boards, talks, art projects, adverts, leaflets, celebrity endorsements and social media.

One subsection considers the effectiveness of awareness-raising about peatlands in general e.g. their biodiversity and value for humans. This subsection also includes education about simple actions to prevent damage to peatlands, such as using peat-free compost and avoiding products containing palm oil, which might develop markets for sustainably produced peatland products. A second subsection considers education about preventing wild fire (e.g. Adinugroho *et al.* 2005). People are a major cause of wild fire on peatland – through arson, carelessness or losing control of prescribed burns. A third subsection considers biosecurity to prevent the spread of problematic species, including the creation of 'black lists' or 'alert species' to which particular attention should be paid.

Related interventions: raise awareness through engaging volunteers in peatland management or monitoring (Section 15.2); provide education/training programmes about peatlands (Section 15.3).

Adinugroho W.C., Suryadiputra I.N.N., Saharjo B.H. & Siboro L. (2005) *Manual for the Control of Fire in Peatlands and Peatland Forest*. Wetlands International Indonesia & Wildlife Habitat Canada, Bogor.

15.1.1 Raise awareness amongst the public (general) Ⓑ Ⓕ Ⓢ

- **One study** examined the effect of interventions to raise general public awareness about peatlands on knowledge, behaviour, peatland habitats or peatland vegetation. The study reported effects on unspecified peatlands.

- **Behaviour change (1 study):** One before-and-after study in the UK[1] reported that following awareness-raising activities, the percentage of the public buying peat-free compost increased.

A before-and-after study in 1990–2007 in the UK (1) reported that following multiple public awareness campaigns about peat in compost, the proportion of people buying peat free compost increased. These results were not tested for statistical significance. In 2007, 35% of people surveyed had purchased peat free compost, compared to 0% before campaigning in 1990. In 2007, 60% of people surveyed were aware of peat free composts and 47% said that it was very or fairly important that their compost is peat free (no data reported for 1990). The 2007 survey, of 1,811 people, was carried out by a UK do-it-yourself retailer. The study does not report the source of the 1990 data. Awareness-raising was carried out by the Peatlands Campaign Consortium, a group of 10 UK conservation organizations aiming to protect peatlands and increase public awareness of their value and degradation. Specific activities included publishing reports and leaflets, organizing seminars, establishing a National Bog Day and placing education volunteers in garden centres.

(1) Alexander P.D., Bragg N.C., Meade R., Padelopoulos G. & Watts O. (2008) Peat in horticulture and conservation: the UK response to a changing world. *Mires and Peat*, 3, Article 8.

15.1.2 Raise awareness amongst the public (wild fire) Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect of interventions to raise awareness about wild fire on knowledge, behaviour, peatland habitats or peatland vegetation.

15.1.3 Raise awareness amongst the public (problematic spp.) Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect of interventions to raise awareness about problematic species on knowledge, behaviour, peatland habitats or peatland vegetation.

15.2 Raise awareness through engaging volunteers in peatland management or monitoring Ⓑ Ⓕ Ⓢ

- We captured no evidence for the effect of engaging volunteers to manage or monitor peatlands on knowledge, behaviour, peatland habitats or peatland vegetation.

Background

Volunteers may be engaged in projects from practical management to citizen science monitoring. Projects that actively engage volunteers to manage or monitor peatlands could indirectly increase awareness of peatlands and their value to the public, change public perceptions towards peatlands and create a sense of ownership over peatlands (Evely *et al.* 2011). Volunteer activities should be carried out sustainably, for example minimizing impact by trampling (Chapter 7) and employing biosecurity measures to prevent the introduction of non-native species.

Related interventions: other interventions to raise public awareness about peatlands (Section 15.1). The effects of specific interventions done by volunteers are considered in the relevant section (e.g. control of problematic species in Chapter 9 and introduction of peatland vegetation in Chapter 12).

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

15.3 Provide education or training programmes about peatlands or peatland management

Ⓑ Ⓕ Ⓢ

- **Two studies** examined the effect of peatland education/training programmes on knowledge, behaviour, peatland habitats or peatland vegetation. Both studies were in tropical peat swamps.
- **Behaviour change (2 studies):** One study in peat swamps in Indonesia[1] reported that over 3,500 households adopted sustainable farming practices following workshops about sustainable farming. One before-and-after study in peat swamps in Indonesia[2] reported that a training course on rubber farming increased the quality of rubber produced by local farmers.

Background

This section considers the effects of education programmes, training courses or workshops, generally aimed at people who manage peatlands (landowners, land managers, local people). They may be about peatlands in general (e.g. their wildlife, their value to humans) or about management techniques (including sustainable land use practices). They may be specifically about peatland vegetation or about broader aspects of peatland ecosystems.

As an example, the 'Bogathon' and 'Sphagathon' programmes in the UK involved a range of land managers (from private estates, landowner organizations, conservation organizations and water companies) in discussions about the most desirable state for upland bogs and how to achieve it (BASC n.d.). In tropical peat swamps, blocking drainage canals can ultimately reduce fire risk, restore forest and create aquaculture ponds. Discussions with local people may alleviate concerns over the immediate loss of transport routes and ensure the blockages remain in place (Page *et al.* 2009).

Related interventions: adopt voluntary agreements or pay landowners to protect peatlands, because these schemes are often linked with education or training (Sections 14.3 and 14.4); allow sustainable use of peatlands, possibly supported by education/training programmes (Section 14.6).

BASC (n.d.) *Grouse shooting and management in the United Kingdom: its value and role in the provision of ecosystem services*. The British Association for Shooting and Conservation.

Page S., Hosciło, A., Wösten H., Jauhiainen J., Silvius M., Rieley J., Ritzema H., Tansey K., Graham L., Vasander H. & Limin S. (2009) Restoration ecology of lowland tropical peatlands in Southeast Asia: current knowledge and future research directions. *Ecosystems*, 12, 888–905.

A study in 2008 in peat swamps in Indonesia (1) reported that workshops with local people encouraged 3,540 households to adopt sustainable farming practices. The study suggests this is a result of changed attitudes towards sustainable farming (but this was not quantified). Workshops were held to identify agricultural and aquacultural practices suited to the local environment but with minimal negative (or even positive) environmental impacts. The workshops involved farmers, government officials, non-governmental organizations, state research institutions and academics.

A before-and-after study in 2010–2013 in peat swamps in Indonesia (2) reported that training local rubber farmers increased the quality of the rubber they produced. No statistical tests were carried out. Once the training was completed, farmers were able to produce rubber with 53% dry rubber content, compared to 45% before the course began. Dry rubber content is a measure of quality, and the higher quality rubber produced after the course fetched higher prices. Between 2010 and 2013, farmers in seven villages received training in rubber farming techniques and economics. The aim was to change the farmers' knowledge and behaviour, so they produced higher quality rubber, made more money from their existing plantations and had less incentive to cultivate remaining peat swamp forests. Details of the rubber quality measurements were not reported.

(1) CKPP (2008) *Provisional Report of the Central Kalimantan Peatland Project*. November 2008.
(2) KFCP (2014) *Practical Lessons from the Field: A Synthesis of Eight Lessons Learned Papers from the KFCP REDD+ Demonstration Activity*. Kalimantan Forests and Climate Partnership, Indonesia.

15.4 Lobby, campaign or demonstrate to protect peatlands Ⓑ Ⓕ Ⓢ

- **Two studies** examined the effect of lobbying/campaigning/demonstrating for peatland protection on knowledge, behaviour, peatland habitats or peatland vegetation. Both studies reported effects, on unspecified peatlands, of the same campaign in the UK.
- **Peatland protection (2 studies):** Two studies in the UK[1,2] reported that the area of protected peatland increased following pressure from a campaign group.
- **Behaviour change (1 study):** One study in the UK[1] reported that following pressure from a campaign group, major retailers stopped buying compost containing peat from important peatland areas and horticultural companies began marketing peat-free compost.
- **Attitudes/awareness (1 study):** One study in the UK[1] reported that following campaign pressure, garden centres and local governments signed peatland conservation agreements.

Background

Lobbying or peaceful demonstrations could put pressure on projects that threaten peatlands, preventing them from occurring or minimizing their impact. This section considers campaigns targeted at organizations such as businesses or governments. Specific actions include demonstrating on site, writing letters and social media campaigns.

The Peatlands Campaign Consortium was formed in 1990 by 10 UK conservation organizations. Its overall aim was to protect UK peatlands of conservation importance, with a focus on (a) on reducing peat extraction and use and (b) raising awareness of the importance of peatlands and the threats they face. Campaigning involved meetings with businesses, parliamentary debates, publishing reports and leaflets, organizing seminars, establishing a National Bog Day and placing education volunteers in garden centres (Rawcliffe 1998).

Related interventions: raise public awareness about peatlands (Section 15.1).

Rawcliffe P. (1998) *Environmental Pressure Groups in Transition*. Manchester University Press, Manchester and New York.

A study in 1993 of peatlands in the UK (1) reported that following a campaign involving multiple individual events, one new area was protected for conservation, seven large businesses changed their purchasing and marketing behaviour to reduce peat extraction, and over 300 organizations signed voluntary agreements to protect peatlands. Within three years of campaigning, protection was granted to 365 ha of peatland (with protection of another 1,134 ha in discussion). Four major retailers stopped buying compost with peat mined from protected areas. Three horticultural companies began marketing non-peat compost alternatives (e.g. coconut fibre compost). Voluntary peatland conservation agreements were signed by 250 garden centres and 51 local governments. The study qualitatively reports some other changes in behaviour, attitudes and awareness. The campaign was run by the Peatlands Campaign Consortium, whose activities included meetings with businesses, debates with governmental organizations, and public awareness-raising (see Background section).

A study in 2008 of peatlands in the UK (2) reported that following pressure from the Peatlands Campaign Consortium, a major peat extraction company donated 3,000 ha of peatland to the English governmental nature conservation body. Campaigning began in 1990 and the peatland was donated in 1992. The campaign was run by the Peatlands Campaign Consortium, whose activities included meetings with businesses, debates with governmental organizations, and public awareness-raising (see Background section). The study does not report how the campaign was related to the donation.

(1) Barkham J.P. (1993) For peat's sake: conservation or exploitation? *Biodiversity and Conservation*, 2, 556–566.
(2) Alexander P.D., Bragg N.C., Meade R., Padelopoulos G. & Watts O. (2008) Peat in horticulture and conservation: the UK response to a changing world. *Mires and Peat*, 3, Article 8.

Appendix 1: List of searched journals/reports

This appendix lists 110 journals/report series searched by the Conservation Evidence project for evidence relevant to the Peatland Conservation synopsis. A further 120+ journals/report series have been searched by Conservation Evidence, but we have not listed them here as their scope is less relevant to the Peatland Conservation synopsis (e.g. with a zoological or marine focus). All issues within the given years have been searched. Some references from sources not on this list have been included in the synopsis when recommended by the advisory board or identified during the summarizing process.

The most relevant journals to the synopsis (i.e. those that contributed papers) are in bold.

Journal Name	From	To
Acta Oecologica	1990	2016
African Journal of Ecology	1963	2016
Agriculture, Ecosystems & Environment	1983	2016
Agroforestry Systems	1982	2016
Ambio	2000	2016
Annual Review of Ecology, Evolution and Systematics	1970	2016
Applied Vegetation Science	1998	2016
Aquatic Conservation: Marine and Freshwater Ecosystems	1991	2016
Aquatic Ecology	1968	2016
Austral Ecology	1977	2016
Basic and Applied Ecology	2000	2016
Biodiversity and Conservation	1994	2016
Biodiversity Science (生物多样性)	1993	2014
Biological Conservation	1981	2016
Biological Invasions	1999	2016
Biology Letters	2005	2016
Boreal Environment Research	1996	2016
Brazilian Biodiversity (*Biodiversidade Brasileira*)	2011	2016
Brazilian Journal for Nature Conservation (*Natureza & Conservação*)	2003	2016
Canadian Field Naturalist (formerly Ottawa Naturalist)	1987	2016
Canadian Journal of Forest Research	2013	2016
Caribbean Journal of Science	1961	2016
Chilean Journal of Natural History (*Revista Chilena de Historia Natural*)	2000	2016
Chinese Journal of Ecology (生态学杂志)	1982	2016
Community Ecology	2000	2016
Conservation and Landscape Planning (*Naturschutz und Landschaftsplanung*)	2003	2004
Conservation Biology	1987	2016
Conservation Evidence	2004	2016
Conservation Genetics	2000	2016
Conservation Letters	2008	2016
Cunninghamia	1981	2016
Ecological Applications	1991	2016
Ecological Management & Restoration	2000	2016
Ecological Restoration	1981	2016
Ecology	1936	2016
Ecology Letters	1998	2016
Écoscience	1994	2016
Ecosystems	1998	2016
Environmental Conservation	1974	2016
Environmental Evidence	2012	2016
Environmental Management	1977	2016
European Journal of Wildlife Research (formerly *Zeitschrift für Jagdwissenschaft*)	1955	2016
Evolutionary Ecology	1987	2016
Evolutionary Ecology Research	1999	2016
Fire Ecology	2005	2016
Forest Ecology and Management	1976	2016

Journal Name	From	To
Freshwater Biology	1971	2016
Functional Ecology	1987	2016
Global Change Biology	1995	2016
Global Ecology and Biogeography	1991	2016
Human Wildlife Interactions (formerly Human Wildlife Conflicts)	2007	2016
Hydrobiologia	2000	2016
iForest	2008	2016
International Journal of the Commons	2007	2016
International Journal of Wildland Fire	1991	2016
Invasive Plant Science and Management	2008	2016
Iranian Journal of Applied Ecology (مجله علمی پژوهشی اکولوژی کاربردی)	2012	2016
Israel Journal of Ecology & Evolution	1963	2016
Japanese Journal of Conservation Ecology (保全生態学研究)	1996	2016
Japanese Journal of Ecology (日本生態学会誌)	1954	2016
Journal for Nature Conservation	2002	2016
Journal of Applied Ecology	1964	2016
Journal of Ecology	1933	2016
Journal of Environmental Management	1973	2016
Journal of Forest Research	1996	2016
Journal of General Biology (*Zhurnal Obshchei Biologii*)	1972	2016
Journal of Mountain Science	2004	2016
Journal of Natural Environment	2010	2016
Journal of Negative Results: Ecology & Evolutionary Biology	2004	2016
Journal of Tropical Biology (*Revista de Biologia Tropical*)	1976	2016
Journal of Tropical Ecology	1986	2016
Journal of Wetlands Ecology	2008	2012
Journal of Wetlands Environmental Management	2012	2016
Journal of Wildlife Management	1945	2016
Land Degradation & Development	1989	2016
Management of Biological Invasions	2010	2016
Mires and Peat	2006	2016
Natural Areas Journal	1992	2016
New Journal of Botany	2011	2016

Journal Name	From	To
NeoBiota	2011	2016
Northwest Science	2007	2016
Oecologia	1969	2016
Oikos	1949	2016
Oryx	1950	2016
Pacific Conservation Biology	1993	2016
Plant Ecology (formerly Vegetatio)	1948	2016
Polish Journal of Ecology	2002	2016
Population Ecology (formerly Researches on Population Ecology)	1952	2016
Preslia	1973	2016
Rangeland Ecology & Management (formerly Journal of Range Management)	1948	2016
Restoration Ecology	1993	2016
Russian Journal of Ecology	1993	2016
South African Journal of Botany	1982	2016
Southwestern Naturalist	1956	2016
The American Naturalist	1867	2016
The Environmentalist	1981	1988
The Rangeland Journal	1976	2016
Trends in Ecology & Evolution	1986	2016
Tropical Conservation Science	2008	2016
Tropical Ecology	1960	2016
Tropical Grasslands	1967	2010
Weed Research	1961	2016
West African Journal of Applied Ecology	2000	2016
Western North American Naturalist	2000	2016
Wetlands	1981	2016
Wetlands Ecology and Management	1992	2016
Wildlife Society Bulletin	1973	2016

Report Series	From	To
Aliens: The Invasive Species Bulletin	1995	2013
Centre for Evidence Based Conservation Systematic Reviews	2004	2016
Scottish Natural Heritage Reports	**1980**	**2016**

Appendix 2: Complete reference list

This appendix lists all references summarized as evidence within the Peatland Conservation synopsis. It does not include references used only in background sections.

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