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CHAPTER THREE

Peatland biodiversity and its restoration

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3.1 Introduction

Biodiversity is 'the variability among living organisms from all sources including, *inter alia,* terrestrial, marine, and other aquatic ecosystems, and the ecological complexes of which they are part: this includes diversity within species, between species and of ecosystems' (CBD 1992). Most of the ecosystem services provided by natural peatlands depend ultimately on living organisms, and individual species may directly deliver provisioning and cultural services. In recent years, peatlands have repeatedly been identified by the Ramsar Convention as the most important wetland type for the support of biodiversity and the regulation of natural processes. They also have been singled out for increased attention by both the Convention on Biological Diversity (CBD) and the United Nations Framework Convention on Climate Change (UNFCCC). Nonetheless, the importance of peatlands for biodiversity is still poorly understood among many audiences.

In this chapter, we outline the biodiversity characteristics of natural peatlands and the suitability of different methods for their assessment (Section 3.2), consider how biodiversity is lost and how losses may be quantified (Section 3.3) and explore some implications for the development of effective approaches to the restoration of peatland biodiversity based on the principles of structural-functional ecosystem analysis (Section 3.4).

3.2 Biodiversity in natural peatlands
3.2.1 Characteristics of peatland habitats and species

The process of peat formation means that part of the peatland biota is directly responsible for creating the habitat (Minayeva *et al.* 2008). Mire massifs form distinct habitat patches that may be spatially separated by large distances, and are characterised by:

* high water level and moisture content
* considerable fluctuations of surface temperature
* low oxygen content
* accumulation of toxic substances and absorbed gases
* limited availability of nutrients
* higher acidity than surrounding ecosystems (in most cases).

These conditions create severe restrictions for living organisms, resulting in intense competition for space and nutrients between individuals even if they have different life forms. Peatlands also influence driving factors (water level, microclimate, matter and water balance, gas exchange, etc.) that affect habitat conditions, and thus biodiversity, for non-peatland ecosystems in the surrounding landscape and downstream.

Most species that are permanently associated with peatlands have developed adaptive strategies during the course of evolution (Rydin and Jeglum 2006), and peatland plants have some distinctive features that are independent of their positions in taxonomic classifications. Typical structural and functional features of the vascular plants include high morphological variability, aeration tissues, extraction mechanisms for toxins and special strategies and mechanisms for nitrogen and mineral uptake, such as insectivory and associations with mycorrhizal fungi. They are predominantly long lived, develop slowly and produce few offspring relatively late in their lifetimes. Thus, they generate populations with stable structure and size. Higher animals (vertebrates) usually use peatlands only at certain stages of their life cycles or during particular seasons, but have also developed adaptations such as the resistance of amphibian and bird egg shells to the acidic environment, specific colourings of fur and plumage, parental care strategies, and synchronisation of life cycles with phenological and weather phenomena. Typically, peatlands host relatively few species (on average no more than 15% of local floras and faunas) but highly specialised species predominate.

3.2.2 Characteristics of peatland ecosystems

Natural peatlands are structurally and functionally organised in a unique way that depends on relationships between plants, peat and water at a range of scales, from the immediate locality to the whole mire massif. Locally, the excess water promotes the dominance of mire plants and impedes decomposition of their dead remains, which consequently accumulate as peat. The physical properties of peat enable it to retain and store a mass of water dozens of times that of its structural matrix. Thus, it can support subsequent generations of living organisms through even the longest periods of normal drought for the prevailing climate. The local conditions depend, in turn, on horizontal connections across the mire massif. A major directional influence is exerted by the lateral movement of water, which both affects and is affected by the presence of plants and peat. This makes peatland a unique ecosystem type in terms of the role that biodiversity plays in its maintenance. Living organisms create and maintain specific abiotic conditions which, in turn, support specialist organisms that are both an integral part of, and highly dependent upon, the ecosystem that is formed. Thus, the peatland ecosystem achieves self-perpetuation on a timescale that is at least one order of magnitude greater than the lifespan of any of the individual organisms involved.

Spatial heterogeneity at various levels is a peculiar feature of most peatlands (Figure 3.1). Different elements of the variously scaled mosaics offer different habitat features, exert different environmental influences and host different ecological processes and phenomena. As a result, ecosystem diversity in peatlands may be described at all spatial levels from the peatland system as a whole down to individual vegetation layers, and biodiversity assessment at the ecosystem level can be carried out within each tier of this hierarchy. For example, the Geographic Information Systems (GIS) archive 'Peatlands of Russia' (Institute of Forest Science, Russian Academy of Sciences) can tell us that the country's peatlands comprise more than 20% permafrost (polygonal and palsa) mire, about 30% transition mire, 18% raised bog, 18% fen and less than 14% ridge-hollow and ridge-pool complexes (Vompersky et al. 2005), or that 62% of the total peatland area is treeless, 21% has open woodland and 17% is covered by forest (Vompersky et al. 2011). Both are general descriptions of the diversity of peatlands occurring within northern Eurasia but, because they reflect different approaches and refer to different ecosystem levels, they cannot readily be compared. Essentially, there is no single answer to the question of how the biodiversity values and functions of peatlands might best be represented, and the expedient solution is to select either the method that that is most appropriate to the purpose or several methods so that the results can be compared.
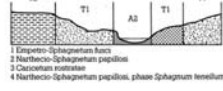

The landscape		Description	Scale (m²)
	Macrotope	The mire complex (or system; several merged mire massifs)	$10^5–10^9$
	Mesotope	The mire massif (separate raised bog, fen, etc.)	$10^2–10^7$
	Microtope	Homogeneous element of landscape heterogeneity within the mire massif (hummock-hollow complex, ryam, margin, sedge mat, *Sphagnum* mat)	$10^2–10^6$
	Microform (nanotope)	Hummock, hollow, pool, hillock	$10^{-1}–10^1$
	Vegetation mosaic	Microcoenosis, tussock, *etc.*	$10^{-2}–10^{-1}$

Figure 3.1 The elements of hierarchical mire classification (after Lindsay *et al.*, 1988 and Masing, 1974).

3.2.3 Supporting biodiversity in other ecosystems

The importance of peatlands for the conservation of biodiversity in other ecosystems arises largely from their environment-forming functions, which operate to support wider biodiversity in various ways, some of which are outlined below.

Support of species from other habitats. Peatlands tend to be the best conserved ecosystems in modern landscapes. These assured quiet zones with comparatively natural habitats provide permanent or temporary refuges for relict plant species and for species at the edges of their ranges, which have been displaced from their original habitats as a result of environmental (including climate) change and/or increasing human impact. They also provide temporary habitats for animal species that use them only intermittently and for particular reasons, or have been forced to move into peatlands from other landscapes. Although most of these animals spend much of their lives in other habitats, they have obligatory relationships with peatlands (Minayeva *et al.* 2008; Minayeva and Sirin 2012).

Support of breeding birds. The peatland avifauna of European Russia comprises some 180 species, of which 146 species belonging to 16 orders breed on peatlands. Relatively few of these are specifically associated with peatlands throughout their seasonal and life cycles. The remainder are less consistently confined to peatlands but, rather, choose them so frequently that peatlands are often their principal regional breeding grounds (V. Nikolaev, pers. comm.).

Stopover sites, feeding stations and short-term refuges for birds. Peatlands play a special role in the support of global flyways. The availability of intact peatlands for staging and feeding on migration routes determines bird population numbers in parts of their ranges that may be distant from their breeding grounds, for example, in Scotland, central Asia or Africa for some of the species that breed in the Arctic (E. Strelnikov pers. comm.).

Ecological networks. Due to their relative naturalness, preservation and stability, peatlands play a key role in the support of landscape connectivity. Watershed and floodplain peatlands, with functional connections via peatlands in intermediate positions within river basins, form a network of 'biodiversity-friendly' habitat that makes them especially valuable for nature conservation. Indeed, the establishment and management of ecological networks in which peatlands function as nodes and corridors is regarded as the most effective approach to nature conservation for densely populated regions. This is especially important under conditions of limited humidity such as those encountered in the steppe and forest steppe regions of Eurasia, as well as in the American prairie. In Europe, the ability of peatlands to support well-preserved habitats and contribute to ecological networks has not been sufficiently exploited in environmental conservation, even though peatlands can be included in regional Natura 2000 Special Protection Area (SPA) systems (Minayeva *et al.* 2008).

3.2.4 Criteria and methods for assessment of the biodiversity status of peatlands

The nature of peatland biodiversity is such that not all assessment methods are applicable. In any other ecosystem type, the energy assigned to storage each year would give rise to a high diversity of ecological niches occupied by different species or forms, all of which would interact to create functionality. In mires the energy is stored as peat, which hardly any species can metabolise, and this in turn limits its expression through habitat diversity. Instead, the energetic potential is realised mainly via intimate biological connections and functionally optimal solutions. Therefore, traditional methods for the assessment of biodiversity status, which are based on structural attributes, are unsuitable for mire/peatland
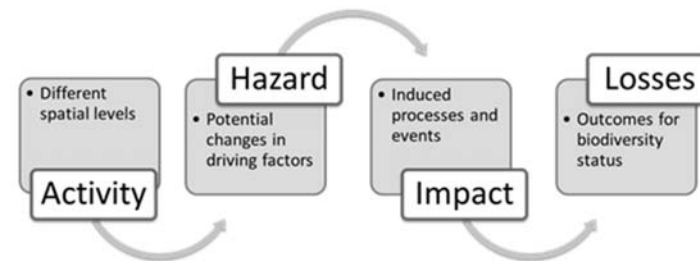

Figure 3.2 The sequence of causality from human activity to biodiversity loss.

ecosystems. Under these circumstances, a functional approach should be applied. Functional effectiveness is often best expressed by the involvement of groups of very small biologically tuned species, such as insects or aquatic invertebrates, which can be used as indicators. Therefore, for peatlands, it is of paramount importance to have an overview of all their components and species and to understand their natural ecosystem processes and functions. Only then might we accurately evaluate their biodiversity status, estimate losses and address these through restoration measures.

3.3 Biodiversity losses

The measures required to restore peatland biodiversity can be identified adequately only on the basis of full information about the causality chain that begins with human activities. These create hazards that may in turn result in impacts leading to biodiversity losses (Figure 3.2). The evaluation of biodiversity losses should follow this chain. It should also be undertaken at different spatial scales, and take account of cumulative effects and biogeographical variability.

The most extensive biodiversity losses are initiated by (macroscale) activities that are applied at landscape level. These activities create hazards such as loss of landscape connectivity and significant changes in climate, hydrology, bedrock, relief, soil (peat), vegetation and species complement which, in turn, impact on natural processes with repercussions that include melting of permafrost, water shortage or flooding, shifts in seasonality, and the disappearance of vegetation cover or even of the peat layer. Some examples of macroscale activities on peatlands are:

- the creation of extensive linear constructions (e.g. roads)
- the construction of large dams and reservoirs
- large-scale opencast mining (e.g. exploitation of oil sands)

- large-scale peat extraction or ploughing for agriculture
- catchment-level overgrazing
- large-scale construction (e.g. airports).

The hazards associated with mesoscale activities, which affect whole mire massifs (Figure 3.1), include shrinkage and compaction of peat. Impacts may be expected across the whole spatial spectrum (microscale to macroscale) and may include changes in hydrology, water level or water quality, the three-dimensional shape of the massif, microtopography, peat thickness and quality, vegetation, species composition and connectivity. Examples of mesoscale activities are:

- drainage or flooding of mire massifs
- small-scale peat extraction
- linear constructions passing through peatlands
- surface pollution and contamination
- small-scale constructions, such as houses
- the conversion of adjacent peatlands into arable land.

Microscale activities alter hydrological factors including water quality, vegetation cover and microtopography. While the primary impacts may occur at microscale, there may be secondary repercussions at mesoscale and above. Microscale activities might include:

- small-scale peat extraction without drainage
- dumping of waste
- pumping in of polluted water
- local water discharge
- construction of recreation facilities including permanent walkways
- industrial berry picking
- lagg or local surface drainage.

As already mentioned, the biodiversity losses arising from impacts at a particular scale may not be restricted to that level in the spatial hierarchy. The relationships between different types of biodiversity loss and the scale of human activities and impacts that may cause them are shown schematically in Table 3.1, and some examples are reviewed in Box 3.1.

Once their nature, origin and potential scale is understood, a quantitative evaluation of biodiversity losses should be within reach. This is important not only for justification and planning of a restoration and/or sustainable management programme, but also to define a baseline condition against

Table 3.1 *The strength of correlation between the spatial scale of impact and type of biodiversity loss.*

Biodiversity losses	Spatial level of human activity and impact		
	Macro	Meso	Micro
Biodiversity of adjacent land and catchments	strong	weak	weak
Mire massif types	strong	weak	weak
Area/variability of mire complex (pattern) types	strong	strong	medium
Diversity of microform patterns	strong	strong	medium
Peat composition types	strong	strong	medium
Present vegetation communities	strong	strong	medium
Productivity	medium	strong	medium
Diversity of habitats	medium	strong	medium
Native species composition	medium	strong	strong
Alien and invasive species composition	medium	strong	medium
Structure of populations	weak	medium	strong
Morphobiology and forms	weak	medium	strong
Genotypes	weak	medium	strong

Strength of relationship between impact and loss:

strong	medium	weak

which the success of restoration measures can be gauged. Like the assessment of peatland biodiversity itself, it is unlikely to follow an existing standard recipe (see Section 3.2.4).

As a general principle, the measurements upon which the evaluation of biodiversity losses is based should reflect real biodiversity characteristics of the peatland in question. Existing practice adopts three distinct approaches to selection of the attributes examined. The first measures structural characteristics and evaluates traditional biodiversity indices (e.g. Fraga *et al.* 2008). The second records functional characteristics and asks how well the ecosystem is working (e.g. Dommain, Couwenberg and Joosten 2010). The third evaluates the socio-economic consequences of biodiversity losses (e.g. Grobler *et al.* 2004). Thus, it is possible to arrive at very different interpretations of a general aim to assess biodiversity-related losses of ecosystem services, depending on the extent of the peatland considered and the focus of the investigator(s). Other treatments (e.g. Page *et al.* 2009) combine all three of the approaches identified.

Box 3.1

From human activities on peatlands to losses of peatland biodiversity: scale changes within the spatial hierarchy

The vegetation or water regime of a peatland may be changed directly by human activities including burning, afforestation, drainage and peat extraction. Because of the close functional linkages between plants, water and peat, any change in one of these components usually affects the others, and impacts need not be restricted to the same level in the structural hierarchy (Figure 3.1). Various authors have observed the consequences of different types of disturbance at individual sites. Some examples chosen to illustrate scale changes are given below.

- Kolomytsev (1993) reports examples from Karelia where small alterations to single components of the plant cover or to the water balance caused dramatic changes in the structure and functioning of the peatland ecosystem, which could lead to complete loss of the mire massif and its associated habitats.
- At Kirkconnell Flow in Scotland, the excavation of a duck pond and a single drainage ditch in the central mire expanse, combined with removal of the uppermost 1–2 m of vegetation and peat from its edges, created conditions that favoured the establishment of self-sown exotic conifer trees across the whole site (Bragg 2004).
- During the first 30–40 years of the twentieth century, the margins of many raised bogs in Scandinavia

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were partially reclaimed and the upper reaches of streams rising there were canalised. Although only a small and peripheral part of each peatland was disturbed, the ecological consequences were far reaching. The modified peatland edges developed uncharacteristically diverse habitats and species complements, the runoff regime was affected and the chemical composition of the streamwater supplied to habitats downstream was altered. Also, habitats on the mire expanse changed as the peat dome began to degrade (Lindholm and Heikkila 2006).
- At Puergschachenmoos, a Ramsar peatland in Austria, there was no evidence of direct disturbance on the mire surface but the vegetation changed gradually over a period of decades. Further investigation showed that the functional peatland unit was much more extensive than the designated area, and the remainder had been converted to agricultural use with concealed drainage (Bragg and Steiner 1995).
- At Clara Bog in Ireland, the excavation of peat from the mire margin (Figure 3.3) caused dramatic subsidence of the peat dome that fundamentally altered its drainage pattern, leading ultimately to changes in vegetation (van der Schaaf 1999).

Figure 3.3 Localised peat extraction at the margin of Clara Bog (Ireland) has caused shrinkage and slumping of the peat body and vegetation change on the mire expanse. This site is now protected as a 'natural mire remnant' (photo: A. Sirin).

3.4 Concepts and methods for peatland biodiversity restoration
3.4.1 Concepts for restoration
When losses of peatland biodiversity have been specified, the possibilities for restoration can be explored. The concept upon which we base our consideration of different approaches is close to that developed by the Society for Ecological Restoration (SER), who define (ecological) restoration as 'the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed' (SER 2004). Defined in this way, restoration encompasses the repair of ecosystems and the improvement of ecological conditions in damaged wildlands through the reinstatement of ecological processes. Such integrative approaches to restoration have been widely adopted over the last decade, and most authors suggest that success should be judged on the basis of an indicator of biodiversity status. The strategy is process oriented, and involves directing autogenic processes while taking landscape interactions into consideration (Whisenant 1999; Van Andel and Aronson 2012). This means that, if the techniques for repairing abiotic factors and processes

that are described in other chapters of this book are implemented effectively, they will also contribute to biodiversity restoration.

In terms of ecosystem dynamics, the likelihood that biodiversity will be maintained or recover after an external disturbance will depend upon whether the disturbance is transient or a long-term chronic pressure and the ability of the peatland to resist, adapt or recover (its robustness, adaptive capacity and resilience) (Dawson *et al.* 2010). Peatlands are equipped with strong feedback mechanisms that, within limits, tend to move the system back towards a stable state after disturbance (Chapter 2). The stratigraphical record suggests that these mechanisms have enabled mire massifs to spontaneously adapt to, and thus to survive through, past changes in climate. In some cases, the same mechanisms may work to move the system back towards an equilibrium state after disturbance caused by human activities. However, because the changes that humans can impose are more abrupt and usually more severe than climatic changes, active restoration work is often needed to assist the recovery, at least if positive results are to be seen within timescales that are relevant in human terms. The need to reverse the effects of human disturbance becomes more pressing when considered against the current backdrop of climate change because the effects are potentially additive, so that reducing pressure from human causes is the best available means to increase the system's capacity to maintain stability as climatic conditions alter (see also the second example in Box 3.2).

The key drivers in selecting an approach for a particular site are statutory (legal) requirements, policy objectives and the availability of funding. Possible practical approaches can be roughly grouped under the five headings below.

3.4.2 Do nothing
Under favourable conditions, the peatland may recover spontaneously after a source of disturbance is withdrawn, but the degree of self-restoration achieved will depend on the situation. This approach has, in effect, been repeatedly adopted through abandonment of peatlands that have been disturbed in various ways, and must always be worthy of consideration for economic reasons. From this point of view, it is obviously expedient to delay active restoration until there is clear evidence of need, and this can only be obtained by collecting sufficient information to identify trends in the unmanaged situation. A robust monitoring programme will be needed. Although often neglected, an initial phase of monitoring is always a worthwhile investment. If the results eventually show that active intervention is required, the initial monitoring phase will provide a baseline against which the success of

restoration measures may be gauged. If, on the other hand, satisfactory progress in spontaneous recovery of ecosystem functions is demonstrated, the considerable expense of active restoration works might be avoided.

3.4.3 Restoration of habitat for populations and species
In some cases, the goal of peatland management is to restore the abundance or population structure of a single target species that has attracted the attention of stakeholders (and funding) because it is rare or endangered. The outcome is usually evaluated on the basis of reproductive success, population size and density, number and variety of individuals, genetic variability or connectivity with other populations. For plants, there are two principal restoration methods. The first reinstates suitable habitats and often relies on natural recolonisation to regenerate the population, but may also involve transplantation. The second involves transplanting specimens of the desired species into existing suitable habitats (Given 1994). For animals, habitat restoration approaches are usually appropriate although reintroduction might be considered in some cases.

This approach to restoration may overlap with restoration of the mire vegetation, the ecosystem, the mire massif or even the landscape. Box 3.2 outlines two cases where conservation requirements for critically vulnerable bird species have indicated needs for peatland ecosystem restoration over large areas. The second example reinforces a recommendation that is common to all available reviews of ecological restoration, namely that species interactions should be taken into account whenever species restoration techniques are applied (Van Andel and Aronson 2012).

On the other hand, there are examples of conservation management for single peatland attributes that negatively affect the natural diversity of the mire ecosystem and/or the biodiversity characteristics of adjacent areas. One is the simulation of historical flax- and hemp-processing activities (Martin and Robinson 2003) by repeatedly excavating pits on bogs for colonisation by *Sphagnum* moss, which promotes the local cover of an important group of mire plants but perpetuates the distortion of natural microtopography and promotes drying out of the surrounding mire surface. Similarly, the reintroduction of grazing or mowing on fen meadows with long histories of traditional extensive management that are no longer required for agriculture (see Chapter 10) may reinstate species-rich 'cultural climax' vegetation, but place a non-natural limit on recovery of the system's peat formation and/or runoff generation functions. Finally, an overriding priority to preserve habitat for the only flock of Taiga bean goose *Anser fabalis fabalis* that winters in Scotland rules out most options for ecological restoration of a flooded peat mine. In cases such as these, where the management of peatland to support single

Box 3.2

Peatland restoration to support vulnerable birds

(a) Aquatic Warbler

The Aquatic Warbler Memorandum of Understanding (MoU) was finalised in Minsk (Belarus) under the auspices of the Convention on Migratory Species (CMS), and became effective on 30 April 2003. It aims to safeguard the globally vulnerable (IUCN Red List) aquatic warbler *Acrocephalus paludicola*. This small migratory songbird was widespread and numerous on European sedge fens at the beginning of the twentieth century but had declined dramatically (by 40% over 10 years) due to drainage of the habitat. Belarus hosts around 40% of the world breeding population

(3000–5500 singing males in 2010). To meet the obligations imposed by the MoU, numerous projects to restore aquatic warbler habitat, mainly on sedge fens in the transboundary Pripyat River Basin (Belarus/Poland/Ukraine), followed. Around 15 000 ha of peatland in Belarus, and similar habitats in Western Pomerania and Poland, were restored within EU 'LIFE' projects (Tanneberger *et at.* 2008) and another 20 000 ha under the auspices of a subsequent German government initiative driven by carbon trading opportunities (Tanneberger and Wichtmann2011).

(b) Golden Plover

The golden plover *Pluvialis apricaria* is a wader that reaches the southern limit of its global range in the UK, where it breeds on upland heaths and bogs. Given the expected poleward shift in species distributions, the UK population is especially vulnerable to climate change (Pearce-Higgins and Green 2014). One potential problem is climate-related decline of its main food species, the cranefly *Tipula paludosa.* Pearce-Higgins *et al.* (2010) have demonstrated a negative correlation between golden plover numbers and August temperature, with a 2-year lag, which is explained as follows. Adult craneflies emerging from the surface layers of peat in May and June can provide a super-abundance of food for

breeding birds, and more golden plover chicks fledge in years when craneflies are plentiful. Granefly larvae suffer high mortality when the surface layers of peat dry out in hot weather. Consequently, in the following year, few adult craneflies emerge and few chicks survive to fledge, resulting in a reduced golden plover population the year after that. This understanding can be used as a basis for developing appropriate management strategies. Because the density of cranefly larvae increases with the moisture content of the peat, the negative effect of hotter summers on golden plover might be reduced by managing water levels on peatlands. Peat wetness could be increased by blocking the drainage ditches (grips) that

were dug across most UK uplands during the last century in a largely unsuccessful attempt to improve the quality of grazing for sheep. Several conservation organisations are already blocking grips for various purposes including amenity improvement and reduction of fire risk, and recent data show that cranefly increase significantly as a result (Carroll *et al.* 2011). This is one of the first studies to show how the resilience

of an ecosystem to climate change might be improved through specific habitat management practices. Importantly, while ditch blocking is already beneficial for peatland conservation, the benefits for birds are likely to increase in the future. This applies not only to golden plover, but also to the wide range of other bird species that feed on craneflies.

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facets of biodiversity may limit the potential for recovery of other ecosystem services, a need for especially clear objective setting is indicated.

For the most severely degraded peat bodies, the rehabilitation approach that is most often applied nowadays involves repair of their structure followed by planting to deliver alternative ecosystem services. Especially if a crop such as cranberries, biomass or timber is produced, these activities may be viewed as another type of species-focused restoration practice. Although they aim to establish non-natural plant communities, and thus to create new ecosystem types rather than to restore natural peatland, some peatland habitat conditions may be retained, e.g. peat soil, shallow water table and low nutrient levels. Paludiculture (wet agriculture) is a refinement which involves cultivating crops of wetland species such as reeds and *Sphagnum* on degraded peatland (e.g. Gaudig *et al.* 2014; Chapter 17). In addition to maintaining peatland ecosystem services such as carbon storage and the delivery of pure water to river systems, some peatland biodiversity value may be regained in conjunction with such commercial uses.

3.4.4 Restoration of peatland vegetation

Much of the biodiversity value of an undisturbed mire massif is concentrated in the surface layer that is occupied by living vegetation (including roots), which is termed the 'acrotelm' in at least some mire types (Ingram 1978). The vegetation itself provides a significant fraction of the system's species biodiversity, and furnishes the three-dimensional habitat mosaic that hosts other life forms ranging from birds and mammals to insects and microbes.

The acrotelm also has a pivotal functional role in maintaining the stability of the mire massif. It receives and partitions rainfall so that, whether or not this is the system's only water source, the peat layer is kept sufficiently wet to prevent aerobic decomposition and ensure that new peat continues to form, the water table remains sufficiently high to support specialised biota and maintain any aquatic elements of the microtopographical mosaic, and water of appropriate quality is discharged to aquatic ecosystems downstream in sufficient quantities and with suitable timing to maintain their biodiversity in turn. Thus, if a degraded vegetation layer is restored, we can expect some recovery in all of these functions. Conversely, if the water regime is restored, there will be benefits for vegetation and thus, again, for other ecosystem functions.

Apart from a few examples of species-focused conservation that intentionally prevent the system from returning to its natural condition (Section 3.4.3), peatland restoration usually aims to promote the re-establishment of self-sustaining natural peatland communities (with associated biodiversity value), even if the policy driver (e.g. water quality, fire prevention, coastal protection) is not biodiversity. The requirements for peatland restoration set by environmental regulators in most countries are rather similar. As a rule, active intervention is expected, aiming at least to achieve the presence of a standard list of species, and at best to restore an appropriate assemblage of habitats. A typical restoration project is conceived as a short phase of intervention that will halt degradation and set the system onto a course of recovery towards a self-sustaining equilibrium condition. This often requires manipulation of one or more abiotic factors such as hydrology, relief, nutrient availability or water quality. Occasionally, full ecosystem restoration has been attempted on very limited areas. Grootjans and Diggelen (2002) identify a whole set of example projects where the management goal 'restoration of vegetation' was achieved by manipulating other ecosystem elements including topsoil, seed and other propagule sources, biomass turnover (via grazing or mowing), water regime and even microclimate (by felling adjacent forest).

Degraded peatlands have usually been drained. Therefore, almost universally, measures to reinstate species and habitat diversity are supported by hydrological manipulations that aim to increase surface wetness. Frequently, drainage ditches are closed in order to raise the water table by retarding the discharge of surface water. The other main approaches are to obstruct (and thus slow down) runoff across bare peat surfaces and in erosion gullies by installing bunds, and sometimes to apply materials such as coir matting, straw mulch or brash, which tend to reduce water loss by evaporation

even if their main purpose is to provide physical protection or support for re-growing vegetation.

Usually, the vegetation is manipulated to directly reinstate mire plant communities. This may involve the removal of undesirable species, such as invading trees on bogs or planted trees on afforested sites (Brooks and Stoneman 1997b; Vitt and Bhatti 2012; Chapter 12). Alternatively, on bare peat where the primary surface has eroded or been removed, desirable species may be introduced by spreading propagules or planting cuttings and seedlings (e.g. Quinty and Rochefort 2003; Carroll et al. 2009; Theroux Rancourt, Rochefort and Lapointe 2009; Chapters 9, 11). Thereafter, imbalanced competitive relationships may be controlled by ongoing vegetation management operations such as sapling removal, mowing or grazing.

In some cases, local microtopography may be adjusted. Ditches on primary mire are usually closed by installing dams which create upstream areas of open water that function as pools. On milled peatland in Canada, pools have been excavated to introduce microtopographical diversity, but their biodiversity was still rather low after 6 years and there may be a need for propagule manipulation (Fontaine, Poulin and Rochefort 2007). Other recent work in Canada has shown that a new *Sphagnum* carpet established on a milled peat surface takes 20 years to develop microstructures comparable to those in natural bogs, and thus to recover ecosystem diversity at this level (Pouliot, Rochefort and Karofeld 2011).

An alternative indirect approach to the restoration of mire vegetation has been adopted for sand-filled oil well platforms in northern Russia and severely eroded peatland in England (Chapter 9). Here, the bare surface is first stabilised by establishing a sward of grasses, with a view to either introducing or allowing natural recolonisation by mire species later. Especially where fertiliser is applied to promote establishment of the grasses, and the grasses are (at least locally) exotic species, the biodiversity benefits may be negative in the initial stages. It is too early to judge longer-term outcomes in general, although the expected replacement of sown timothy grass *Phleum pratense* by a peatland species (arctic cottongrass E*riophorum scheuchzeri*) occurred in just 4 years at one oil well site in Nenets Autonomous Okrug, Russia (A. Popov, unpublished data).

The intensity of propagule supply is important not only for spontaneous re-vegetation, but also for managed restoration. The scientific literature reports many instances of seed rain/propagule shortage constraining the success of restoration projects, and many techniques to overcome this problem have been developed for various habitat types (e.g. Harper 1977;

Rochefort *et al.* 2003; Klimkowska 2008). Good practice for any biodiversity restoration project should include a full evaluation of seed and propagule sources at an early stage. One of the baseline studies for restoration of the stream-valley fen Drentse Aa (The Netherlands) investigated the soil seed bank, the wind-blown seed rain and the seed influx from the coats of animals as well as in their droppings. The results indicated that grazing animals can be used to carry plant propagules into areas undergoing restoration (Grootjans and Diggelen 2002; see also Vander Kloet et al. 2012). Another clever strategy that has been applied on tropical peatlands encourages birds to deposit seed in areas under restoration by installing artificial perches (Graham and Page 2012). For non-peatland ecosystems, the success of restoration work has been enhanced by creating streams to transport propagules (Engstrom, Nilsson and Jansson 2009), and this technique might be considered for peatlands under some circumstances, although hydrological aspects would need very careful attention. There have also been numerous studies of the role of floods in seed dispersal for riparian habitats which may be relevant to peatland restoration, especially for floodplain mires (Jansson *et al.* 2005; Groves *et al.* 2007).

3.4.5 Mire massif restoration

Where attempts to restore peatland vegetation using the methods outlined in Section 3.4.4 have failed, the cause often lies at a higher level of the structural hierarchy. Vegetation can re-establish successfully only if sufficient water of appropriate quality is available at the ground surface. There is no prospect of achieving this if the total annual loss of water from the peat body as a whole, by seepage, exceeds the net supply. Such imbalances can arise if the base area of the peatland has been reduced (e.g. by peat cutting at the margins as illustrated in Figure 3.3), if its hydrological boundary has been altered by peripheral drainage, or if a groundwater supply has been diverted. In such cases, appropriate restoration measures will aim to establish hydrological stability at the level of the mire massif (Bragg 1995).

If the peat body has been severely disrupted, restoration of the original vegetation may no longer be a viable proposition and the best that can be done is to establish an ecosystem type belonging to an earlier developmental stage. For example, if a bog has been cut down to the fen peat layer, fen vegetation may establish more successfully than bog vegetation. At some Canadian sites where peat extraction had exposed minerotrophic (fen) peat, re-vegetation was relatively rapid but important genera (e.g. *Carex* and *Sphagnum* spp.) failed to colonise spontaneously (Graf, Rochefort and Poulin 2008) so that measures to artificially

introduce these key species were still required. If the residual peat layer is very thin and flooding is a problem, lake or swamp may be the only viable target for restoration to a self-sustaining wetland ecosystem. This will at least set a course that could eventually result in establishment of a peat-forming ecosystem.

3.4.6 Landscape approach

In order to realise the full biodiversity potential of a restored peatland, it will be necessary to consider not only the mire massif itself, but also its connections to other similar habitat patches, for example through reproductive and dispersal mechanisms whose ranges vary widely between different peatland species and life forms. If peatlands are too widely spaced within the landscape, recruitment may become impossible for some populations of mire species. Otherwise, population processes occurring under isolated conditions may render them genetically unsustainable. This is the ecological networks concept of interconnectivity, which addresses the need to ensure free movement of wildlife between fragmented habitat patches and may also involve island biogeography theory. Its potential application in the present context is to determine which degraded mire massifs should be afforded the highest priority for restoration in order to achieve a spatial distribution of mire habitat patches within the landscape that is optimal in terms of the interconnectivity requirements of at least the critical characteristic species.

A related consideration is the spatially varying capacity of the physical environment to support peatland systems, insofar as this will influence the degree of correspondence that can be achieved between a practically achievable distribution of mire massifs and the theoretical optimum. This links to the principles of hydro-genetic mire classifications and the extent of peatland losses described in Chapter 2. A legacy of human activities in densely populated areas is that peatlands have disappeared from many physically suitable locations. Their potential extent remains accessible through modelling (e.g. Mclnnes *et al.* 2007; Franzen *et al.* 2012), which may be required to inform any plans to (re)place missing nodes within the habitat-patch network.

3.5 Conclusion

It is clear that the distinctive biological diversity of natural peatland is much more than a species list. Rather, it is the 'top-level' expression of a combination of ecosystem structure and function that has taken millennia to develop and is responsible for delivery of the majority of peatland ecosystem services. It follows that the restoration of peatland biodiversity at any scale

will usually require more than just the reintroduction of missing species. The appropriate management prescription will be site-specific and determined not only by natural site features, but also by the legacy of hazards and impacts created by previous human use. Manipulation of abiotic factors such as hydrology and geomorphology will usually be required, as well as some consideration of climate change effects, and it will almost always take time for the outcome to be fully manifest. Site restoration alone may not be sufficient to achieve the maximum functionality of peatlands as refugia and in providing habitat connectivity to enable the adaptive migration of species in a changing climate. For this, the spatial distribution of mire massifs within the wider landscape will become increasingly important. Given this role of peatlands in supporting species and in the provision of a wide range of other ecosystem services, a broader spatio-temporal approach will be needed to safeguard their full value for biodiversity in the future.

3.6 Acknowledgements