Forestry Commission

Deforesting and Restoring Peat Bogs

A Review

Russell Anderson

Technical Paper 32

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Front Cover: East Flanders Moss, Stirlingshire, two years after tree clearance and drain blocking. The site is very wet and supports good regrowth of hare's-tail cottongrass but the straw-coloured wavy hair grass probably indicates nutrient enrichment.

Back Cover: *Top* Peat cracking eventually reaches a stage at which cracks in plough furrows are joined up by a network of cracks in the peat beneath the tree rows.

Middle Installing a plywood sheet dam in a main drain to help rewet and restore Horse Hill Moss, one of the Border Mires in Kielder Forest. Materials that don't rot, such as plastic piling, are now recommended for such work.

Bottom Given the right conditions, bog vegetation can regrow on clearfelled bogs. Here heather, hare's-tail cottongrass and *Sphagnum capillifolium* are starting to obscure felling debris.

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CONTENTS

Summary

There is a demand for advice on the restoration of afforested bogs. This paper critically examines the evidence that bog restoration can succeed and considers the ecological and cost-effectiveness of bog restoration operations.

Afforestation affects bogs by changing their physical and chemical environment, triggering biological changes as well as affecting the wider environment. Most of the values we attach to undisturbed bogs are adversely affected by afforestation. The primary purpose of bog restoration is to re-create wildlife habitat. There is little evidence that restoring bogs will improve river fisheries or limit climate change by increasing carbon storage.

A high water table is the key to restoring bogs and action to raise the water table is likely to be necessary. There are ecological arguments for retaining plough ridges and furrows, provided their drainage effect can be negated. Trees must be felled or killed to provide sufficient light for re-establishment of bog vegetation.

On very flat sites, blocking drains at key points can rewet large areas, but the areas affected rapidly diminish with increasing slope. Where there is no scope for raising the water level by blocking drains at key points, peat cracking, if sufficiently advanced, may hinder rewetting by allowing water to by-pass dammed up furrows. The installation of a continuous impermeable barrier in a trench may assist rewetting on these sites.

Practical bog restoration has had mixed success. Infilling drains with the original spoil has been successful. Felling trees and damming plough furrows has raised water levels more than either operation on its own. However, there is very little published evidence of vegetation succession on restored bogs because current projects have not been monitored or are too recent. There are some reports of increased *Sphagnum* cover but it is too early to judge if this will lead to successful restoration.

The costs of bog restoration vary substantially with treatment and age of forest. The net cost of clearing trees (i.e. the clearance cost minus any income from the sale of products) decreases as the forest grows. Felling to waste can cost as little as £250 per ha for pre-thicket forest. Treatments that leave the site relatively clear of debris cost at least £400 per ha. Forwarders, skylines and helicopters have been used to harvest whole trees with net costs ranging from £1250 to £9000 per ha. Reported costs of damming drains and furrows range from £6 to £42 per dam for furrows and normal drains but rise to £190 per dam for large drains. The cost of damming plough furrows depends on their gradient, £2000–£4000 per ha at 0.5° rising to £10 000–£15 000 per ha at 2°. Monitoring costs upwards of £500 per year.

Judging cost-effectiveness is difficult because the desired and actual end-points are often unclear; the cost of restoration must take account of the change it brings about in the value of the land and the benefits cannot be expressed in monetary terms. The feasibility and cost-effectiveness of restoration are influenced by the integrity of the bog edges, the flatness of the site, the drainage layout (i.e. the potential to rewet large areas by blocking drains at key points), the presence of remnant bog vegetation in the forest or on adjacent land, the need to build harvesting roads and the availability of brash.

The feasibility of successful post-forestry bog restoration cannot yet be judged. There is insufficient evidence to justify a policy encouraging large-scale bog restoration. Instead, a limited number of monitored projects should proceed, together with further research.

The priorities for further research are: to ascertain the success of current projects, to determine threshold water regimes for bog restoration, to determine the importance of limiting nutrient release, to develop and improve restoration methodology and to evaluate alternative habitat creation options.

Glossary

Acrotelm	The thin, fibrous, upper layer of peat in which the water table naturally fluctuates. It is very much more permeable and has more oxygen available than the underlying peat (the catotelm).
Amorphous peat	Black or dark brown, highly humified peat with few visible fibres, often containing ancient pieces of birch. Found in shallow blanket bogs and in the lower metre of deep blanket bogs.
Blanket bog	May refer to ombrotrophic part of blanket mire but often used loosely in place of the latter.
Blanket mire	Expanse of mire covering both flat and sloping land, found in the uplands and the most oceanic parts of the lowlands. Consisting mainly of bog but also including small areas of fen. Includes FC soil types 8, 9, 10b, 11, 14 (Pyatt, 1982).
Bog	Ombrotrophic mire. Mire whose vegetation only receives nutrients in rainwater or in surface runoff from adjacent bog, resulting in the accumulation of strongly acidic peat, very low in plant nutrients. Includes FC soil types 9, 10, 11, 14.
Bog woodland	Bog supporting sparse woodland, usually of Scots pine, constrained in terms of tree cover and growth rate by the nutrient limitations of the site.
Catotelm	The layer of peat comprising the entire depth of the peat deposit below the bottom of the zone in which the water table fluctuates (the acrotelm). Almost permanently saturated, impermeable and devoid of oxygen. Usually pseudofibrous and/or amorphous in texture but in lowland raised bogs occasionally fibrous and composed of compacted *Sphagnum*.
Fen	Mire whose vegetation receives nutrients not only in rainwater but also from mineral soil or rock via groundwater. The resulting peat is weakly acidic, neutral or alkaline and relatively nutrient-rich. Includes FC soil types 8, 9a.
Forest bog	Bog surrounded by forest and therefore having a more sheltered, shady and frost-prone microclimate than bogs in open country.
Fibrous peat	Peat composed of densely packed but relatively undecomposed plant remains. Varying in composition from tough *Eriophorum* fibres to fluffy, almost raw, compacted *Sphagnum*.
HAP	Habitat action plan.
Intermediate bog	Transitional rather than inherently different type of raised bog showing some tendency to spread over sloping land. Found on the upland/lowland margin where the climate is only just too dry for blanket bog formation. Includes FC soil types 9, 10, 11, 14.
Lowland raised bog	Raised bog in the lowlands.
Lowlands	Used here to mean land less than 250 m above sea level with less than 160 wet days per year (not an agreed definition but one suitable for defining the areas of Britain in which blanket bog does not occur).

Mire	Peat-forming wetland. Includes fens and bogs but not lakes or salt marshes (Moore and Bellamy, 1974).
Natural primary bog	Primary bog dominated by an actively-growing and *Sphagnum*-rich surface pattern. This is the bog condition class (Lindsay, 1995) representing pristine and near-pristine bog.
NVC	National vegetation classification.
Ombrotrophic	Literally rain fed. Receiving nutrients solely from the atmosphere (in rain, snow and wind-borne dust) or in the barely enriched runoff from adjacent ombrotrophic bog.
Peatland	Land with peat soil. Some definitions specify a minimum peat thickness of 20–45 cm (Paavilainen and Päivänen, 1995).
Primary bog	Bog whose surface has never been removed except by natural processes such as fire and therefore contains a stratigraphic record of the bog's development (c.f. secondary bog). Afforested bogs are partly secondary (i.e. the plough furrows) but mostly primary.
Pseudofibrous peat	Peat containing visible fibres among a brown gelatinous matrix of partly decomposed plant remains. The fibres are a distinct but minor component and have lost most of their strength.
Raised bog	Discrete, gently domed, ombrotrophic bog. Raised bogs formed in the uplands have generally become incorporated into blanket mires and the term usually refers to lowland raised bog. Includes FC soil type 10a.
Scandinavian-style peatland forestry	Drainage of peatlands, usually combined with fertilizer application, to increase the growth rate of slow-growing natural forest stands.
Secondary bog	Bog which has been cut over or had its surface vegetation removed at some stage.
Wet woodland	Birch, willow or alder dominated woodlands on wet soils. Group of woodland types comprising NVC types W1–W4a and W5–W7.
UKBAP	UK biodiversity action plan.
UK-style peatland forestry	Plantation forestry on formerly treeless peatlands prepared by drainage, ploughing and fertilizer application.
Uplands	Defined in this context as land over 250 m above sea level or with more than 160 wet days per year. Blanket bog may form in the uplands on flat to moderately sloping land with inherently poorly drained soil.

Chapter 1

Introduction

In the early 20th century, successful but labour-intensive methods were developed for forest planting on bogs (Zehetmayr, 1954; Anderson, 1997). Drainage and cultivation were achieved using manual labour. Fertilizer, in the form of basic slag, was cheaply available as a waste product of iron smelting. By the late 1950s techniques had become sufficiently developed to allow large scale planting of peatlands and, as forestry was largely excluded from the better agricultural land, an increasing proportion of new planting took place on peatland sites. A realisation of the value of undisturbed peatlands as wildlife habitat ended the period of peatland afforestation and since 1990 little further planting has taken place. Of Britain's 2.1 million hectares of non-fen peatland with a peat depth greater than 45 cm, about 190 000 ha, or 9%, has been afforested (Cannell *et al.*, 1993).

As conservation and enhancement of biodiversity have increased in importance, the topic of habitat restoration has gained in prominence. The UK Biodiversity Action Plan, implementing commitments under the Rio and Helsinki Agreements, includes action plans for key habitat types. These include lowland raised bog and blanket bog, both of which are also listed as habitats of community interest in the European Union Habitats Directive. The action plans include targets for areas of habitat to be restored from agriculture and woodland, leading to a demand for advice on the feasibility and desirability of restoring bogs currently planted with forests.

Guidelines for peatland have been published as a Forestry Commission Guideline Note *Forests and peatland habitats* (Patterson and Anderson, 2000). The section of these guidelines covering policy on peat bog restoration is reproduced here as Appendix 1. Practical advice on techniques for bog restoration is given by Brooks and Stoneman (1997a), Wheeler and Shaw (1995), and Foss and O'Connell (1998).

This Technical Paper stems from a joint Forestry Commission / Scottish Natural Heritage study of the scope for deforesting and restoring bogs. It covers all types of bog likely to be encountered in and around UK forests, referring to three broad classes: raised bogs, blanket bogs and intermediate bogs (see Glossary). It does not cover fens. Based on a review of literature and on information from practical bog restoration projects, it aims to:

- provide a background to the subject of restoring bogs after afforestation
- critically examine the evidence that bog restoration after afforestation can succeed
- consider the ecological benefits and cost-effectiveness of restoration operations
- list criteria for predicting sites where restoration is most likely to be cost-effective.

Chapter 2

Why attempt to restore bogs that have been afforested?

The question 'Why attempt restoration?' can be answered by considering the impacts of afforestation on bogs, how they change the various values we attach to bogs and the overall desirability of reversing the changes.

Impacts of afforestation

The impacts of afforestation can be dealt with by considering firstly the ecological changes to the site (i.e. changes in its physical and chemical environment), secondly the biological responses, and thirdly the impacts on the wider environment.

Much of the literature on the effects of draining and afforesting peatlands concerns either drainage for agricultural improvement or Scandinavian-style peatland forestry (see Glossary). To make use of this knowledge, the findings of relevant studies have been considered in the context of UK-style peatland afforestation (see Glossary).

Physical and chemical changes to the site

Impacts of blanket bog afforestation on water levels, peat nutrient status and microsite conditions were reviewed in the context of their influence on the feasibility and ease of bog restoration by Anderson *et al.* (1995). Most of the impacts also apply to raised and intermediate bogs. The ecological effects of peatland drainage for forestry, particularly Scandinavian-style and including North American experience, were reviewed by Laine *et al.* (1995a). These and other ecological impacts can be summarised as:

i. Discontinuation of light burning and grazing regimes used before afforestation can change ground conditions by allowing more litter from *Calluna vulgaris, Eriophorum angustifolium, Molinia caerulea* and *Deschampsia flexuosa* to accumulate, thickening the layer of vegetation, plant litter and raw peat (Shotbolt *et al.*, 1998) and shading out many other species including some Sphagna (Chapman and Rose, 1986,1987). Smith and Charman (1988) suggested an alternative mechanism for a change in vegetation of unplanted bogs within forests, a gradual drying out caused by the surrounding forest, but a follow-up study (Charman and Smith, 1992) showed vegetation composition to be more strongly related to peat depth than to distance from the forest edge.

ii. Forestry drainage (i.e. plough furrows and deeper intercepting ditches) causes a slight lowering of the water table relative to the ground surface (Anderson *et al.*, 1995) with an associated subsidence of the surface due mainly to compression of the peat (Anderson *et al.*, 2000). The efficiency of drainage in lowering peatland water tables is higher in low rainfall areas (e.g. Hillman, 1987) and lower in very high rainfall areas (e.g. Coulson *et al.*, 1990). However, forests in high rainfall areas would be expected to dry out the peat (see v. below) after canopy closure. Ploughing creates a long-lasting micro-relief of drier ridges and wetter furrows. The furrows continue to provide damp, humid hollows after the natural hollows in the bog surface have dried out.

iii. Shade and shelter increase gradually as the trees grow and cover more of the ground, increase markedly around the time of canopy closure and then stay fairly constant until harvesting, when both are reduced dramatically.

iv. Plant nutrient element concentrations in the upper 50 cm of the peat profile increase with time in ombrotrophic bogs and remain unchanged with time in mesotrophic mires following drainage (Laiho and Laine, 1994). Applying fertilizer to a forest growing on an ombrotrophic bog increases the rate of nutrient cycling by increasing nutrient concentrations in needle litter (Finér, 1996).

v. Interception loss and transpiration by trees leads to a much greater lowering of the water table than is caused by drainage alone (e.g. Binns, 1959) and a greater surface subsidence (Pyatt *et al.*, 1992) due to both shrinkage of the drying surface layer and compression of the underlying peat (Shotbolt *et al.*, 1998; Anderson *et al.*, 2000). The hydraulic conductivity of the dried layer increases while that of the compressed layer, the peat below the water table, decreases (Ivanov, 1981).

vi. Large scale peat cracking associated with shrinkage occurs as a stage of the drying process (Pyatt and John, 1989). The first cracks in blanket bog peat can appear after the trees close canopy at 10–15 years and can develop rapidly (Pyatt *et al.*, 1987). The cracks eventually form a network (Binns, 1968; Howell, 1984) which effectively acts as an additional drainage system and may hamper attempts at restoration. Pyatt and John (1989) found that fibrous peat, such as the relatively undecomposed upper 5–20 cm typical of blanket bogs, does not crack, but cracking occurs below this depth and has also been observed on many raised bogs.

vii. Clearfelling causes the water table to rise (e.g. Berry and Jeglum, 1988; Päivänen, 1974), although in one study (Verry, 1981) clearcutting led to more extreme seasonal fluctuations in water level, conditions unlikely to favour rehabilitation of a bog. Studies of peat compression in a civil engineering context (Berry and Poskitt, 1972; Edil *et al.*, 1986; Hobbs, 1986; Samson and La Rochelle, 1972) suggest that a small amount of rebound (i.e. rise) of the surface may occur following the relief of overburden pressure (e.g. by tree removal and / or a rise in water level), but it is unlikely to be significant.

Biological responses

Changes to the physical and chemical environment affect plant and animal communities, altering the species composition and the production rate. Microbial communities are also affected and changes in the activity of different functional groups can slow down, or even reverse, the rate of peat accumulation.

Vegetation change

Irrespective of the direct effects of trees the composition of bog vegetation will change after taking land into forestry management because burning and grazing regimes involved in the previous land use are discontinued. Ground which is not otherwise disturbed (i.e. by drainage, ploughing, tree planting or fertilization) can suffer a reduction or loss of some bottom layer species including Sphagna, and an increase in abundance of some field layer species (Table 1 (a)). *Sphagnum* cover can also increase as a result of reduced trampling and burning and *Betula nana* can be released from deer suppression (Table 1(a)).

Drainage on its own has little effect on the vegetation of blanket bogs, changes being confined to a narrow strip alongside the drains and starting to disappear after 10 years (Table 1 (b)). The drainage, ploughing, fertilizing and planting used to establish forests on UK bogs combine to alter the vegetation quickly and dramatically (Table 1 (c)). The relatively undisturbed ground between plough ridges has its water table lowered, to the benefit of heather and other ericaceous shrubs, but the application of fertilizer favours the cottongrasses (Anderson *et al.*, 1995). Many of the bog species that are tolerant of wet conditions and low nutrient levels lose their competitive edge in the improved conditions created to help the trees grow (Usher, 1996).

As a tree stand develops on a bog, the ground vegetation gradually changes from a bog type to a forest understorey type (Table 1 (d)). The most gradual change occurs where woodland spreads over relatively undisturbed bog passing through a stage intermediate between bog and forest. Pioneer Scots pine woodland, still with ground vegetation characteristic of raised bog, may succeed to mature woodland with *Calluna* shaded out more than *Eriophorum vaginatum* (Longman, 1995). Scandinavian-style peatland afforestation, involving drainage of bogs with sparse natural stands of Scots pine, usually results in an abrupt increase in the cover and biomass of field layer dwarf shrubs and heath-coniferous wood mosses and a more gradual decrease in the cover of most *Sphagnum* species (Kurimo and Uski, 1988; Gustafsson, 1988; Vasander *et al.*, 1993; Laine *et al.*, 1995a, 1995b). Fertilizing these drained sites also reduces *Sphagnum* cover and increases the cover and biomass of *Eriophorum vaginatum* (Kurimo and Uski, 1988). In general, drainage leads to an increase in the availability of plant nutrients (Holmen, 1964). Post-drainage vegetation succession is most rapid in the more fertile mire site types than in the poorer ones, because higher nutrient levels allow more rapid tree stand development (Laine and Vanha-Majamaa, 1992).

Although UK-style peatland afforestation starts off with treeless sites, stand development is very rapid because of the ploughing or mounding, fertilizing, close tree spacing and favourable climate, so vegetation change here will usually be more rapid than in Scandinavia. Once the canopy has fully closed the ground flora is reduced to sparse *Eriophorum vaginatum*, occasional ferns, a few mosses, and some liverworts and lichens. The orchid *Listera cordata* (and possibly also *Goodyera repens*) can become locally common in lodgepole pine (*Pinus contorta* Douglas ex Loud.) plantations on blanket bog.

Second rotation forests will be different from first rotation ones in two aspects. The forest will have been re-designed during the first rotation felling / restocking stage. Open ground will have been redistributed within the forest and mires will

Table 1. Vegetation changes resulting from (a) the discontinuation of burning and grazing, (b) drainage alone, (c) a combination of drainage, ploughing, fertilizing and planting, and (d) tree stand development

	Bog type and location (reported by)	Species increased	Species decreased
(a)	Intermediate bog, Northumberland. 30 years since afforestation (Chapman and Rose, 1986, 1987)	*Mylia anomala*	*Drosera rotundifolia* *Narthecium ossifragum* *Sphagnum magellanicum* *Sphagnum papillosum* *Sphagnum capillifolium* *Odontoschisma sphagni*
	Blanket bog, Sutherland (A. Coupar, personal communication)	*Sphagnum* spp. *Betula nana*	
(b)	Blanket bog, Pennines (Stewart and Lance, 1983)	*Calluna vulgaris* (beside drains only)	*Sphagnum* spp. (beside drains only)
(c)	Blanket bog, Caithness 3 years since afforestation (Anderson *et al*., 1995)	*Eriophorum vaginatum* *Calluna vulgaris* *Erica tetralix*	*Trichophorum cespitosum*
	Blanket bog, Caithness 7 years since afforestation (Usher, 1996)	*Eriophorum angustifolium*	*Trichophorum cespitosum* *Erica tetralix* *Drosera rotundifolia* *Narthecium ossifragum* *Sphagnum papillosum* *Sphagnum capillifolium* *Racomitrium lanuginosum* *Odontoschisma sphagni* *Cladonia portentosa* *Cladonia uncialis*
(d)	Raised bog, Peeblesshire (Longman, 1995)	*Deschampsia flexuosa* *Eriophorum vaginatum*	*Calluna vulgaris*
	Sparsely wooded raised bog, Sweden (Holmen, 1964)	*Hylocomium splendens* *Pleurozium schreberi* *Dicranum* spp.	*Sphagnum* spp.
	Raised and blanket bogs, UK sites. 20–60 years after planting (Anderson, unpublished data)	*Hypnum jutlandicum* *Plagiothecium undulatum* *Dicranum* spp. *Rhytidiadelphus* spp. *Listera cordata*	*Calluna vulgaris* *Erica tetralix* *Eriophorum vaginatum* *Eriophorum angustifolium* *Trichophorum cespitosum* *Molinia caerulea* *Sphagnum* spp.

often be unplanted, even if they were planted over in the first rotation. Their vegetation composition will depend on whether action has been taken to conserve or restore a mire community and on its success. Where no action has been taken, grassland, heath or scrub communities may develop. The second major change will be a progressive loss of the original plant communities and an increase in woodland flora in parts of the forest which are replanted.

Invertebrate community change

Soil-living invertebrates are affected less by drainage of peatlands than those that are surface-active. Most groups of the former benefit from drier soil conditions (Vilkamaa, 1981) perhaps because their food supply improves. Earthworms also increase in species richness (Makulec, 1991). If soil conditions become too dry, most soil dwellers can simply move from hummock to hollow (Markkula, 1982) or migrate deeper. Surface-active invertebrates do not have this option because they are not adapted to below-ground living. Their habitat may be more severely affected than that of soil dwellers because the vegetation structure and composition is altered, affecting temperature, moisture and light conditions. Drainage of a raised bog in south-west Finland caused no net change in spider numbers, species richness or diversity, but the composition of the community changed. Typical bog species were lost while forest species and less habitat-specific bog species moved in (Koponen, 1985). The degree of invertebrate community change may be related to that of vegetation change. At a Pennines blanket bog site, where drainage had triggered a vegetation change from heather to grass, numbers of herbivores (Hemiptera, Diptera, Elateridae, Staphylinidae) increased and those of some predatory species of ground beetles and spiders decreased. At another site where drainage had been less effective, fewer changes occurred (Coulson, 1990).

A far greater impact on surface dwellers than that of drainage alone can be expected from afforestation because it involves both drainage and a rapid change from open bog to forest conditions. Five years after afforestation of blanket bog in Caithness, numbers of surface-active beetles, moths, plant bugs, harvestmen and slugs had increased while wasp and spider numbers had decreased (Coulson, 1990). The species composition had changed markedly. Two upland heath indicator species and one upland blanket bog indicator had increased while all three lowland mire indicators found on adjacent undisturbed bog had declined. Exclusion of deer and sheep from the unafforested bog had little effect on the invertebrate fauna.

Bird community change

Birds that use open peatlands, some of which rely heavily on peatland habitats for breeding or obtaining food, suffer an abrupt loss of habitat after afforestation. Some, such as golden plover (*Pluvialis apricaria*) can tolerate ploughing and drainage of the ground and others, such as short-eared owl (*Asio flammeus*), may benefit temporarily from the increase in vole numbers. However, once the trees are established, the change from open ground to forest is too extreme for most open ground birds and they are displaced. Some are able to use the forest edge but not the interior and some can use open ground within the forest. The merlin (*Falco columbarius*) sometimes nests in trees in the forest. It may use bog pools in openings in the forest to catch dragonflies for a short spell in summer but continues to need open ground to hunt for food. Snipe (*Gallinago gallinago*) live and breed on unplanted bogs within forests while teal (*Anas crecca*) sometimes breed on dubh lochans in these bog enclaves.

Forest birds start to use the habitat as it develops. Foliage and bark insects provide a substantial food source, as does tree seed once it becomes available. Nesting habitat safe from most predators is provided by the trees. Forest songbirds themselves become an important food supply for raptors capable of hunting among trees, such as sparrowhawk (*Accipiter nisus*) and goshawk (*Accipiter gentilis*).

None of the forest species is unique to peatland forests. Osprey (*Pandion haliaetus*) often nest in sparsely wooded bogs but plantations rarely provide the combination of seclusion and suitable forest structure (i.e. occasional tall trees among stunted ones) that they favour. Wood sandpiper (*Tringa glareola*) may also be dependent on sparsely wooded bogs in forests.

Mammal community change

Laine *et al.* (1995a) reviewed literature on the effects of Scandinavian-style peatland forestry on mammals. Some findings are relevant to the UK. Herbivores may become more abundant because drainage causes an increase in food and shelter. Field vole (*Microtus agrestis*) and common shrew (*Sorex araneus*) increased after forest drainage (Hansson, 1978). Increased use by moose (*Alces alces*), of peatland sites drained for forestry, as wintering areas and nutrient-enriched feeding grounds, may not be paralleled in UK by deer because they are normally initially fenced out of afforested ground.

Some mammals, such as weasel, fox and pine marten, prey on voles. Their numbers often increase, presumably in response to a growing vole population in the first few years after afforestation, then decline as the forest canopy closes (Moss *et al.*, 1996). Wildcat numbers generally increase in proportion to the amount of margin between open country and woodland (Moss *et al.*, 1996). Woodland mammals, such as bats and roe deer may benefit from peatland afforestation.

Forests usually contain substantial areas of open ground. Pyatt (1993) used as an example the Forest Enterprise forests in Caithness and Sutherland, which had 50 sq km unplanted out of a total area of 400 sq km. Although the closed canopy forest may be of low quality as habitat for most mammals, these areas continue to provide food and shelter for herbivores and presumably continue to support increased densities of their predators.

Impacts on the wider environment

Edge effect of peat drying and subsidence on adjacent land

Peat dries, subsides and cracks beneath forests in response to lowering of the water table (Pyatt *et al.*, 1987). In first rotation forests, only a narrow strip of adjoining unplanted bog is affected. The width of this strip increases as the forest grows. For South Coastal lodgepole pine on blanket bog in Caithness the width was 10–20 m at age 19 (Pyatt *et al.*, 1992), 30 m at age 28, and was expected to be about 40 m by harvesting time at age 35 (Shotbolt *et al.*, 1998). The width of the affected zone is expected to increase in second and subsequent rotations if the effects of the first rotation are not reversed after harvesting, because the drying process will, in the next rotation, restart where it left off.

Predator risk to ground nesting birds on adjacent land

There is evidence that unplanted peatland adjacent to conifer plantations can become less useful as breeding habitat for some moorland birds. Stroud and Reed (1986) claimed to have demonstrated that some breeding densities of curlew, golden plover, dunlin, lapwing, snipe and redshank were affected up to 800 m from the forest edge; greenshank were unaffected. However, the validity of these results is in doubt because the analysis used to obtain them was shown to have been flawed (Avery, 1989; Lavers and Haines-Young, 1993). Parr (1992) studied red grouse and moorland waders on blanket bogs in Caithness, comparing densities, breeding success and evidence of predation among the three land categories: afforested; unplanted but adjacent to forests; unplanted moorland distant from forests. Higher numbers of predators were seen on open ground near forests than distant from them. There was some evidence of an associated higher rate of predation on adult red grouse, though not enough to affect overall densities. Density and breeding success of both red grouse and waders were unaffected by adjacent forests up to 8 years old. Older forests appeared to cause a reduction in the density of some wader species and in the breeding success of red grouse and golden plover on adjacent open ground.

The findings of these and other reports on bird predation near forest edges have been summarised by Moss *et al.* (1996). After afforestation, some open ground predators (e.g. raven, red kite, golden eagle, hen harrier, merlin and short-eared owl) tend to be replaced by forest species (e.g. goshawk, sparrowhawk and tawny owl). Others continue to use the land after afforestation (e.g. stoat, weasel, pine marten and fox). Some forest edge predators (e.g. crow, magpie, kestrel, long-eared owl, buzzard and wildcat) increase in numbers in proportion to the length of forest edge/open country boundary. Sparse evidence suggests that most effects of afforestation on predation extend to about 1 km from the forest edge. One study found that curlew and snipe densities were higher next to forests, possibly because vegetation tended to be taller near forests.

In summary, golden plover and possibly dunlin appear to be adversely affected by adjacent forests, but there is a lack of firm evidence for other effects.

Streamflow and water quality impacts

Jones (1987) reviewed the literature on how streamflows are affected by forestry drainage and afforestation within deep peat catchments. Annual runoff increases for a period of a few years following drainage, then decreases again, possibly ending up lower than before drainage. The increase seems to be not only the result of extra runoff generated when water is released as the water table is lowered. The secondary compression process resulting from a lowering of the water table occurs over a period of a few years and involves water being squeezed out of the saturated peat below the water table. Runoff begins to decrease again when the increase in evapotranspiration from the growing trees reaches a level which matches the decreasing rate of water yield of the compressed peat. Peak flows have been reported to increase for 1–5 years or to decrease due to an increased water storage capacity in the drained layer. Spring or summer peak flows increased in many cases, even when there was an overall reduction in peak flows. Peatland drainage might increase a stream's effective catchment area. Evidence of effects on base flows has been conflicting, but on balance it appears that drainage increases base flows by providing a larger water store from which runoff is generated in dry weather. More recent results from the Coalburn study in Kielder Forest (Robinson *et al.*, 1998) have confirmed that peatland drainage increases annual runoff and low flows and enhances peak flows. Forest growth reduces water yields and peak flows but does not cancel out the drainage-induced enhancement of low flows. For the first 20 years or so the dominant influence is that of the drainage; thereafter the effect of the trees becomes equally important.

Changes in the values we attach to bogs

In the UK, bogs are valued for several reasons. They contribute to upland Britain's unique barren landscapes. They provide variety in predominantly agricultural landscapes. They provide wildlife habitat, particularly scarce in the agricultural lowlands. They play a role in regulating global climate by storing carbon. They supply rivers with unpolluted, low nutrient content, low ionic strength water. They provide a means of studying the climate of the past in order to develop ways of predicting, and possibly influencing, future climate and they preserve archaeological information and artefacts. Table 2 summarises the effect of afforestation on these values.

It is clear that productive use of bogs as forestry land has an effect on the wider values. Of the values listed in Table 2, it is the likely net decrease in their value as wildlife habitat which has prompted the strongest opposition to further afforestation of peatlands and led to calls to remove forests and restore bogs.

It may be possible to reverse most of the value changes in Table 2 by deforesting and restoring bogs. Chapter 3 examines some of the arguments for doing so.

Table 2. How the values of bogs are affected by forestry

Value	Change in value on afforestation
Economic	Increase
Barren landscape	Decrease
Variety in farmed landscape	Increase or no change
Nature conservation	Usually net decrease
Climate regulation by storing carbon	Uncertain, probably little change
Source of high quality water	Decrease
Palaeoclimatic archive	Decrease
Archaeological archive	Decrease

Chapter 3

The case for restoration

Nature conservation

Bog ecosystems

Raised bogs have a wide geographic range but have become rare both in western Europe and in the British Isles due to exploitation. Peat extraction, agricultural drainage and afforestation have drastically reduced the area of active raised bog. Only 5% of Britain's former 70 000 ha of raised bogs is now in near-natural condition (Lindsay and Immirzi, 1996). In most countries of north-west Europe the situation is similar. Active raised bog is recognised to be an endangered habitat in Britain and in the European Union. The EU Habitats and Species Directive makes its conservation a statutory responsibility of member states (EC, 1992). The UK Habitat Action Plan (HAP) for lowland raised bog includes among its proposed targets the identification by 2002 of areas, timescales and targets for restoration or improvement of significantly altered raised bog areas, including those used for woodlands. A further HAP proposed target is the initiation, by 2005, of management to improve or restore these areas.

Blanket bogs are far more extensive in Britain than raised bogs, occupying perhaps 2 million ha (Lindsay, 1995), but are scarce in Europe and in the world as a whole (Lindsay, 1987), Britain's share comprising around 17–20% of the world resource (Tallis, 1995). The majority of Britain's area of blanket bogs is primary (i.e. uncut) and active (i.e. supporting a significant cover of vegetation that is normally peat-forming). Small-scale peat cutting for domestic fuel has removed the primary surface in areas near roads, but the practice of replacing the turf at the foot of the peat bank allows good natural regeneration, producing an active secondary surface similar in many respects to primary bog. Afforestation has curtailed bog growth in some areas, the surface remaining largely primary, but peat formation ceasing as the surface dries out. Lindsay (1995) stated that the remaining primary active bog has been degraded by overgrazing, burning, moor draining and the impacts of acid rain, to such an extent that only 10% was estimated to survive in a natural state. It is important to remember that man's impact in the past has helped shape what we now think of as natural bogs. Some may have been created or prevented from succeeding to woodland through past land use.

The UK blanket bog Habitat Action Plan recognises four condition classes: favourable; degraded but readily restored; degraded but less readily restored; and degraded and probably beyond restoration. The Plan has four targets for the period up to 2015. Firstly, it aims to maintain the current extent and overall distribution of blanket mire currently in favourable condition. Secondly, it aims to improve the condition of the areas which are degraded but readily restored to bring the total area in, or approaching, favourable condition to 340 000 ha by 2005. Thirdly, it aims to introduce management regimes to improve to, and subsequently maintain in, favourable condition a further 280 000 ha of degraded blanket mire by 2010. Lastly, it aims to introduce management regimes to improve the condition of a further 225 000 ha of degraded blanket mire to, or approaching, favourable condition, by 2015. The Plan does not define the condition classes, although it does acknowledge that many afforested areas may be too degraded to merit restoration.

Britain's area of intermediate bogs was estimated by Lindsay and Immirzi (1996) to be about 12 000 ha, most of which had been affected by afforestation, moor draining, burning, overgrazing, peat extraction or opencast coal mining. In Kielder Forest, most have been left unplanted, or have only had their margins planted. A few have been completely planted (Lowe, 1993). The conservation value of this type of bog may lie in the richness of its flora and fauna; it can sustain elements of both the upland and lowland bog biota. However, as part of the continuum from lowland raised bog to blanket bog they contain no unique species of their own.

Species

Cotton deer-grass (*Trichophorum alpinum*) provides an example of a species unique to bogs which has become extinct from the British Isles. It was last found on Restenneth Moss in Angus but has not been found since the bog was drained (Clapham *et al.*, 1987). Rare or threatened bog-dependent species which are present, perhaps still surviving in unplanted areas within the forest (e.g. *Sphagnum balticum*), might justify bog restoration.

Improving forest biodiversity

The Forestry Commission's forest design planning process is intended to make forest managers think carefully about how to distribute open ground within the forest to achieve the greatest environmental benefits. Landscape naturalisation (i.e. management aimed at producing a more natural landscape) has been recommended as an appropriate means of enhancing biodiversity in Britain's upland spruce forests (Ratcliffe and Peterken, 1995). The natural forests of Britain, Scandinavia and the Pacific Northwest are suitable reference points for mimicking natural features; all contain open mires. The biological composition and structure of mires surrounded by forest may be different from those of mires on open moorland and this 'forest bog' habitat may have distinct biodiversity values justifying restoration of former bogs, even in areas with an abundance of open blanket bog.

Several peatland habitats other than open or forest bogs are, in certain circumstances, appropriate endpoints for restoration. Bog woodlands and transition zones at planted forest edges adjoining open blanket bog may add valuable structural variety to forests. Research is needed on the potential value of these habitats and on the feasibility of, and techniques for, creating them. Some more nutrient-rich peatlands naturally support wet woodland, usually dominated by birch, willow or alder. Former bogs which have been markedly enriched with nutrients from fertilizer applications may have the potential for restoration to wet woodland.

Carbon sink and greenhouse gases

Bogs form the largest terrestrial stores of organic carbon. Afforestation of formerly treeless bogs causes further carbon to be sequestered in the trees but at the same time, probably reduces the peat carbon store by drying the surface layer sufficiently to enhance microbial breakdown of organic matter and release carbon dioxide. These two processes counterbalance one another but it is thought that in UK conditions a net release of carbon dioxide to the atmosphere occurs (Cannell *et al.*, 1993). Emission of the more powerful greenhouse gas, methane, which is released by bogs in their natural state, seems to stop when afforestation takes place and it may be that forested bogs actually remove methane from the atmosphere by increasing the activity of methane oxidising bacteria in the aerated upper peat layer. One study reported that rewetting a bog previously drained for forestry caused a big increase in late summer methane emissions but that the annual emission rate remained lower than that of similar sites which had never been drained (Komulainen *et al.*, 1998).

It is not yet possible to judge the overall effects of peatland afforestation and restoration on atmospheric concentrations of greenhouse gases. As well as carbon dioxide and methane, nitrous oxide concentration may be affected

Fisheries

It has been claimed that river fisheries can be improved by restoration of blanket bog. This may be an extrapolation from the possibility that afforestation of blanket bogs has adversely affected them but there is no evidence that bog restoration will benefit fisheries. In areas with acid-sensitive surface waters, it is likely that removal of forest will benefit water quality by reducing the input of scavenged air-pollutants. Conversely, on shallow blanket bog, where forestry drainage or ploughing has cut through to the mineral soil, afforestation may have improved the buffering capacity of stream water (Ramberg, 1981). In these circumstances, rewetting the site may reduce the mineral influence, so that streams become more vulnerable to acidic episodes.

In some circumstances restoration can reduce ground-water quality. Hughes *et al.* (1996) reported that bromide was released into soil water by decomposing organic matter from a flushed bog in Wales, which had been rewetted by piping streamflow from an adjacent gully on to the surface.

Forestry returns

Unlike in Finland, where at least 10% of the forestry-drained peatland area is considered not worth harvesting (Vasander *et al.*, 1992), Britain's more favourable climate and our use of ploughing and planting to afforest peatland sites have ensured the generally successful establishment of forests here. Nevertheless there are areas within our forests where tree growth has been poor and either the trees are not worth harvesting or the site is not worth restocking. Some of these areas are peatland sites where initial drainage, drain maintenance or fertilizer inputs have been inadequate, or heather competition has not been controlled. Others may be inherently unsuitable for growing trees, e.g. small basins with a high water table and little scope for drainage. In both cases bog restoration is one of the management options available and is an attractive one because it will contribute to achieving UKBAP Habitat Action Plan targets. The decision whether to replant must be based on an assessment of the potential to restore bog habitats on the site and the cost of doing so, as well as on the economics of further site amelioration for restocking.

Chapter 4

Basic principles of restoration

The aim: ecosystem functions or a specific community?

Current bog restoration projects usually recognise the uncertain feasibility of restoring the former vegetation. The Border Mires project (Burlton, 1996) gave priority to maintaining and enhancing open mires, rather than regaining any specific vegetation community. The Stell Knowe restoration project aimed to restart peat formation, rather than restore a particular vegetation type (Stoneman, 1994; Longman, 1996). Projects in Finland have aimed to regain a functioning mire ecosystem, not a specific type of mire or mire vegetation (Heikkilä and Lindholm, 1995b). The Langlands Moss rehabilitation project was perhaps alone in setting out to demonstrate the feasibility of specifically restoring a raised bog (Brooks and Stoneman, 1997a, 1997b).

Wheeler and Shaw (1995) discussed the options for cut-over raised bog restoration objectives. Much of their discussion was relevant to post-forestry bog restoration too. They underlined the need to consider aiming to restore an earlier developmental stage than the ombrotrophic bog which immediately preceded peat cutting (or in our case afforestation). The desirability of conserving any remaining wildlife interest needs to be taken into account alongside the desirability and feasibility of restoring a fen or fen woodland community as a precursor to eventual bog development.

Is it necessary to raise the water level to restore a bog?

A water table close to the surface is required to support bog vegetation. In considering the necessity of taking action to raise the water level, two questions arise. What level is required to ensure a succession to bog vegetation and to what level does the water table rise after tree felling?

The water table in undisturbed bogs generally lies within 10 cm of the surface during the winter but falls a further 10 cm or more in summer. It is assumed that unless the water table can be returned to these levels, the drier conditions will not favour restoration of a bog habitat but there is no scientific evidence on the matter.

When a forest is felled the water table rises initially (e.g. Päivänen, 1980; Pyatt *et al.*, 1985) because the tree canopy is no longer there to intercept and re-evaporate a proportion of the rainfall and the tree roots stop taking up water from the peat. If the trees are removed from the site, the removal of their weight may allow some rebound of the ground surface but any rise in level is likely to be very small and may not affect the distance between the water table and the surface. The water table may not rise to its pre-forestry level because the drainage system still exists and the peat in the rooting zone may have become more permeable while the trees were growing. The water table may rise further subsequently if the drains deteriorate due to sedimentation or vegetation growth.

There is plenty of anecdotal but little scientific evidence that open drains can become blocked up by vegetation growing in them. *Sphagnum cuspidatum* and *Juncus bulbosus* are good colonisers of standing water in drains. *Sphagnum recurvum* and *Eriophorum angustifolium* often grow on drain bottoms. Although drains can, in time, become completely infilled by vegetation, it is not known how much of a blockage to water flow this represents. Bare peat on exposed drain sides can be eroded by frost action and the resulting deposit in the drain bottom can gradually reduce the effective depth of the drain but probably not sufficiently to rewet the bog. Drains which have sufficient water flow or are overhung and shaded by heather do not become colonised by vegetation and do not deteriorate.

Is it desirable to minimise nutrient concentrations?

It would seem so, given that an implicit objective of most bog restoration projects is to regain a cover of ombrotrophic vegetation. During afforestation the site will have received nutrients in the form of fertilizer. Mineralisation of dried peat beneath the forest may have released further nutrients, which may have been taken up by the trees. Thus the nutrient capital of both site and vegetation will have increased since the start of the afforestation process. If foliage or whole trees are left on site during restoration, a proportion of the nutrients contained in the trees will be released by decomposition following harvesting. This could allow the site to support a more nutrient-demanding type in place of the former bog vegetation. Colonisation of bog vegetation by grasses, particularly *Molinia caerulea*, is regarded as a sign that the successional direction is away from bog and towards more nutrient-demanding grassland, scrub or woodland communities.

Reports that fertilizer additions can help the artificial establishment of *Sphagnum* on bare peat surfaces after peat extraction (Rochefort *et al.*, 1995; Ferland and Rochefort, 1997) probably have little relevance to post-forestry bog restoration. It is unlikely that peat on a clearfelled forest site would be as nutrient poor as the peat exposed by peat cutting.

Is it necessary to remove lop and top from the site?

Any opportunity taken to remove nutrients from the bog, for example by removing tree foliage or whole trees from the site at the time of felling, will, in theory, increase the chances of re-establishing ombrotrophic bog vegetation. This has not yet been confirmed in practice but is a high priority for research, given the additional cost of removing foliage from sites.

Besides the risk of adding nutrients to the site from decomposing tree foliage there may be other consequences of leaving lop and top on site during restoration. Shelter and weed suppression effects of retaining harvesting residues on site were detected in a study of the initial growth of planted second rotation forest (Proe *et al.*, 1994). On a bog restoration site the residues might reduce tree natural regeneration by shading a proportion of the ground but presumably tree seedlings which do appear would receive the same benefits as the planted seedlings in the Proe *et al.* (1994) study. It is difficult to predict how the two effects would balance out. *Sphagnum* can also benefit from the partial shade and shelter afforded by felling residues, sometimes becoming the dominant ground cover in the shaded areas, in contrast to the unshaded ones (personal observation at Longbridge Muir).

Are there advantages in not felling the trees?

To restore a site to anything approaching a natural bog ecosystem the forest canopy must be largely removed because its shade prevents the survival of non-shade-tolerant species among the ground vegetation. This can be done by felling the trees or by killing them *in situ*.

Shade-tolerant bryophytes predominate on the forest floor once the canopy has closed. Several *Sphagnum* species (i.e. *S. cuspidatum, S. recurvum, S. capillifolium, S. papillosum, S. palustre, S. fimbriatum)* are capable of surviving for some time in loose, open clumps or within the litter layer after canopy closure. These surviving plants may be valuable sources of material for revegetation when restoration is being attempted. Because, when growing under a forest canopy, their growth form is more open, there is a risk that exposing them suddenly to full sunlight and wind could dehydrate and destroy them. A more gradual change in conditions of shade and shelter might increase their chances of survival. This provides some justification for killing trees standing but perhaps an equally gradual change could be accomplished by leaving felled whole trees or harvesting residues (i.e. branches and tops) on site.

It seems probable that techniques such as girdling, poisoning or prescribed burning exist or could be developed for killing trees at a lower cost than that of felling them. Against this must be weighed the undesirability, in terms of landscape and amenity value, of having a forest of dead trees standing on the 'restored bog' and the risk of rooks colonising the skeleton trees and causing nutrient enrichment of the site. Past bog restoration attempts have usually involved felling the trees because it has been considered unacceptable to have stands of dead trees on the restored bog.

Will blocking the drains kill the trees?

Lodgepole pine and Sitka spruce have shown no sign of dying or of their growth slowing two years after the plough furrows were dammed at an experiment in Caithness, even though the site is undoubtedly wetter than before. It seems that the rewetting has not been sufficient to kill the trees but there remains the possibility that it may stress them, making them more vulnerable to defoliation by pine beauty moth in future.

Is it necessary to restore the whole of a bog?

Forest Design Plans and the Woodland Grant Scheme usually aim for a 10–20% open ground component in the forest. In the case of forests wholly confined to large lowland raised bogs (e.g. West Flanders Moss, Foulshaw Moss, the Lochar

Mosses) this encourages attempts to restore small parts of the bog. Hydrological theory shows that a lowered water table on part of a bog leads to lowering of the water level over the whole bog. It has been claimed that bog restoration should therefore apply to all or nothing, that unless the whole bog is tackled, hydrological functioning cannot be restored so effort spent on partial restoration is wasted.

The hydrological theory is probably correct but the consequence misinterpreted. The amount of lowering of the water table over the whole bog may be very small and not enough to prevent active bog growth. The survival of many of the specialist bog plants has been observed in the ride network of some first rotation forests on bogs (Anderson, 1998, 2000). These bog remnants have undoubtedly changed from their pre-afforestation condition but not so markedly as to have led to the loss of many bog species. The existence of these refugia encourages a more pragmatic approach where whole-bog restoration is not practicable. 'Restoration' of areas within the forest, which retain some of the functions and most of the species of the former bog, seems likely to be a worthwhile interim measure while we await the results of research on the feasibility of bog restoration. Open ground in the lowland raised bog forests should be distributed in areas centred on parts of the ride network with the best remnant bog vegetation and should be managed to enlarge those remnants by restoring bog adjacent to them.

Is it necessary to flatten the ploughing or fill in drains?

Plough ridges and furrows give clearfelled forest sites a highly unnatural appearance. This is undesirable in bog restoration projects because early evidence of success is perceived in a renewed naturalness of the site. Levelling the ploughing would get rid of the stripes. It might also help to raise the water table. Unfortunately it is not easily done. Unless the trees are very young, their roots will have grown through the ridges into the underlying peat, making it difficult to turn the ridge back into the furrow. Some plough ridges on the restored, formerly forested Langlands Moss (Lanarkshire) have eroded rapidly in the first six years since restoration (personal observation). The ridge peat looks likely to completely erode away within 10 years in some areas, leaving the tree stumps and roots which will take longer to rot down. Brooks and Stoneman (1997) reported the infilling of unplanted forestry plough furrows by the RSPB at Abernethy Forest. The ploughed ground had apparently not been planted and Hymac diggers were successfully used to turn the ridges back into the furrows and compact them down.

The ecological effects of the ploughing may be less drastic than the visual effects. Their drainage impact is obvious and can be tackled as discussed in Chapter 6. The furrows provide wet or damp, slightly sheltered hollows which, to some degree, mimic the wet hollows found in a natural bog surface. These hollows will be valuable regardless of how effective any rewetting treatments are. If the surface remains dry they will provide relatively damp conditions and if it rewets they will provide shallow pools. Research on methods of restoring bogs which have been abandoned as flat expanses of bare peat following peat extraction have demonstrated the value of ridges and furrows for establishing *Sphagnum* from fragments (Ferland and Rochefort, 1997). The ridges may provide hummock-like microsite conditions. Ridges and furrows have also been used to increase microtopographic variation in an attempt to restore an otherwise flat cut-over peat surface at Leegmoor in northern Germany (Wheeler and Shaw, 1995, p. 96).

Sphagnum can rapidly fill wet hollows and this process is often seen within a few years of clearfelling forests on bogs, even when no action is taken to raise the water table. It is likely that plough furrows will quickly become invisible if bog vegetation can be restored.

Drains may need to be blocked by damming to reduce their effect on water levels in the peat. However, the resulting deep water-filled channels can be extremely hazardous when walking on the site. The danger is often increased because they become overgrown with heather or other vegetation and so are practically invisible. Deep drains are likely to remain hazardous for many years because of the slow rate of occlusion. The importance of this hazard will depend upon who has access to the site. In some cases it may be staff only and they can be made aware of the danger. In most cases, however, members of the public are likely to be at risk. The hazard could be reduced or avoided by filling the drains in with peat. It is important to distinguish between drain infilling just to make the site safer and drain infilling to raise the water level. The latter will require working to a higher specification to ensure efficacy.

Chapter 5

The special case of cracked peat

It has been suggested that cracking of peat beneath plantations may hinder attempts at restoration by forming a secondary drainage system less easily blocked than the forestry drains and plough furrows (Anderson *et al.*, 1995). A good example exists on part of a bog restoration experiment in Braehour Forest, Caithness (personal observation). The peat is severely cracked in part of the site where there is a slope of 5° and the peat is less than one metre thick. The plough furrows have been dammed with plastic piling inserted down to the base of the peat but are not ponding up any water, even in winter, presumably because it drains away via the cracks.

How widely will this problem arise?

Pyatt (1987) described three stages of cracking occurring in forests on deep blanket bog. Cracks sometimes first develop along the base of deep ditches, followed within a few years by cracks along plough furrows (Plates 1 and 2) and several years afterwards by the formation of a network of vertical cracks in the undisturbed peat between furrows (see Back Cover).

The rate of shrinkage is rather variable, depending on peat depth, degree of humification, tree species and climate. It is difficult to generalise about the plantation age at which cracking begins but it has been observed in a raised bog beneath a 9-year-old lodgepole pine stand (Pyatt, 1976). It is more usual for the first cracks to be seen shortly after canopy closure, perhaps 10–20 years from planting. The process may reach the crucial third stage in a plantation aged 20 years or older.

Cracking can be expected on all blanket bogs under any tree species including deciduous ones (Pyatt *et al.*, 1987). Pyatt and John (1989) reported that the fibrous peat characteristic of the upper 10–20 cm of blanket bogs does not crack; however the peat underneath this upper fibrous layer does. On blanket bog with vigorously growing trees, cracking will develop to form a network within about 20 years of planting. It is not known whether the

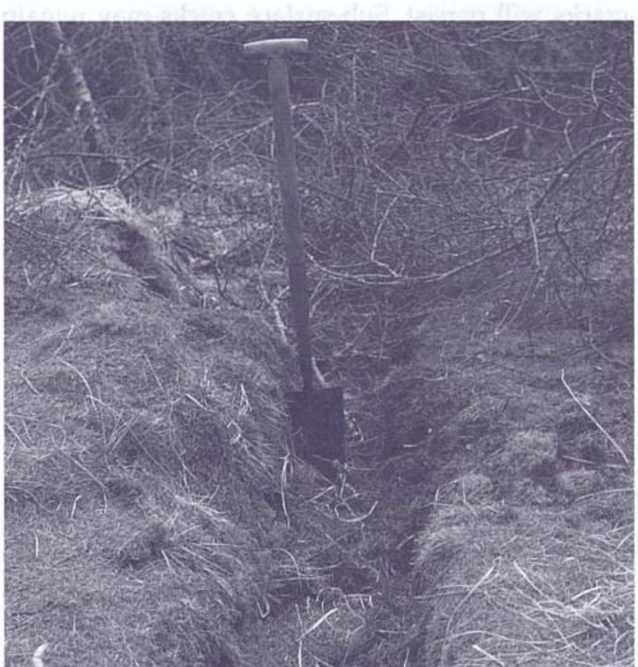

***Plate 1** Cracks in the base of plough furrows are not evident on the surface because they are covered by accumulated litter.*

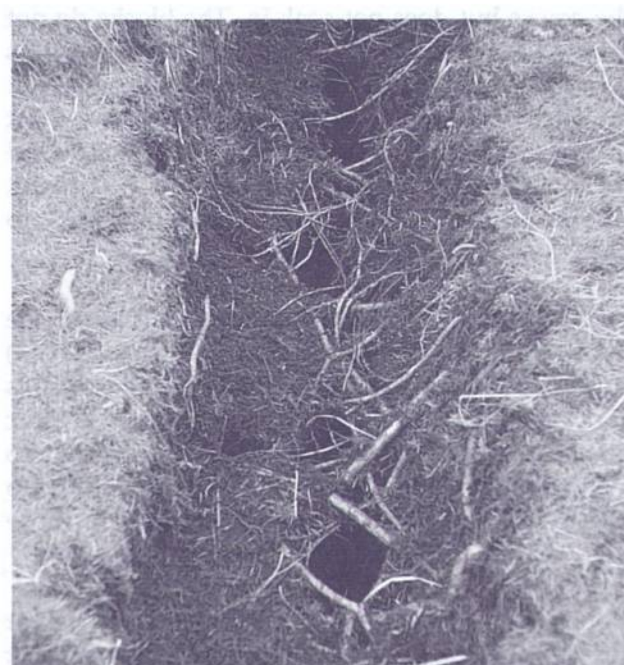

***Plate 2** Cracks in furrows become visible when loose litter and the tough root and litter mat are removed.*

almost undecomposed, fibrous, *Sphagnum* peat found in some lowland raised bogs, is subject to cracking. Cracking will occur on any type of bog provided that non-fibrous peat exists within the zone where sufficient drying occurs, the upper one metre of the peat profile. Thoroughly cracked peat will probably be found in all bog restoration sites where plantations have reached maturity. Some raised bogs with very fibrous peat may be exceptions.

Severe cracking will not always cause problems for restoration. Rewetting of some very flat sites will be achieved simply by blocking drains at key points where they cross rides or roads. Cracking does not occur in unplanted peat so the roadside or ride acts as a barrier to water moving through the crack network. If cracked peat beneath the forest is rewetted in this way, the cracks may remain open but water will not drain away. The high water table will support bog vegetation so that a layer of new peat can begin to form above the cracked layer.

Reversibility of cracking

For centuries peat has been used as a domestic fuel and the shrinkage process which accompanies drying is well known. Peat for household use, usually amorphous to pseudofibrous in texture, is dried at the peat bank and taken home only when the blocks have hardened and their weight and volume decreased substantially. It is stacked outside, usually without any cover. When it is wetted by rain, water may lie on the surface or in the cracks but does not soak in. The blocks do not swell, demonstrating that the shrinkage is irreversible. Pseudofibrous peat harvested as an industrial fuel is also stored uncovered because it dries irreversibly. Air-dried fibrous peat is not used for fuel, because its density is too low. It does not shrink much on drying and is probably prone to partially rewetting.

There have been few experimental studies of the reversibility of peat shrinkage. Haines (1923) studied the shrinkage and swelling of mineral soils and noted that some soils with a high organic matter content did not swell when subjected to wetting after drying. Gentle oven-drying appears to cause irreversible changes analogous to those produced by firing clay.

Hobbs (1986) reported that partially dried peat, at any stage of the process, could not recover the lost moisture on re-submergence, explaining permanent material change in the peat as due to oxidation. Cores of fibrous fen peat subjected to partial oven drying and then submerged in water re-absorbed some water and expanded but neither the water content nor the volume returned to the original condition. These results show that for some types of fibrous peat the drying and shrinkage process is only partly reversible but it is not certain whether this applies to the raw *Sphagnum* peat found in some lowland raised bogs.

During winter, cracks in peat beneath plantations fill with water if the drainage ditches are not sufficiently deep to remove it (Pyatt, 1987). The cracks remain open in spite of these prolonged periods of waterlogging, indicating that cracking, like the shrinkage which causes it, is irreversible.

Will cracks infill naturally?

We have observed various stages of cracking at many sites and found the following general features. In the early stages of cracking, the layer of needle litter hides cracks in the bases of ditches and furrows (Plate 1). Fine tree roots permeate the more humid lower part of this layer, forming a tough mat. Only when cracking has reached the third stage do the ditch and furrow cracks become visible. By this time they have become so wide that the root and litter mat tears. The network of cracks between plough furrows does not open on the surface because the more fibrous surface peat seems less prone to cracking and is reinforced by strong structural tree roots (see Back Cover).

There is no scientific evidence on how long the cracks will persist. Sub-surface cracks may remain permanently because the peat shrinkage process is largely irreversible and they are protected from frost and vegetation colonisation. The existence of apparently long-lived natural fissures in deep peat confirms that this is possible, although water flow may be a factor in their longevity. Open cracks might also persist or might gradually fill in due to frost erosion and / or vegetation colonisation.

Importance of slope

Post-felling water levels in thoroughly cracked peat will depend on whether water is able to escape from the cracks. Bands of uncracked, relatively wet peat along rides and roadsides prevent water crossing these except in the acrotelm or in drains. Subsidence of the ground surface under the plantation relative to these unplanted areas may limit water removal from the cracks, but in some cases exit drains will have been deepened as part of a drainage maintenance programme.

On extremely flat sites, the potential exists to rewet or even swamp large areas by blocking key exit drains where they cross rides or roads. However, extremely flat sites are rare and on more sloping sites the area affected by blocking key drains will be modest. In most cases some other method of raising the water level will be needed to restore sites with cracked peat.

Techniques for rewetting bogs with cracked peat

Inoue *et al.* (1992) described the use of a waterproof membrane installed in a narrow trench to raise the water level in a bog in a Japanese National Park. Thirty metre long rolls of 1.2 m wide, 0.3 mm thick vinyl sheet, were installed in a 550 m long trench. The 15 cm wide trench was dug using a machine designed for controlling water loss from rice paddies, adapted by reducing ground pressure and fitting rubber tracks, so that vegetation damage was prevented. Insufficient water level data were presented to demonstrate success or failure.

There is potential for developing a similar technique for rewetting cracked peat. An alternative of infilling the trench with wet peat, tamped down to form a relatively impermeable barrier, has the potential to reduce the cost. Another alternative, using a heavy vehicle to disrupt and compact the surface by driving across it, also deserves trial.

Prospects of restoring sites with cracked peat

Sites where peat cracking has advanced to the third stage, in which the cracks form a secondary drainage network, may present problems if attempts are made to rewet them in the course of bog restoration. Very flat sites should not prove problematic because it should be possible to rewet them simply by blocking drains where they cross rides and roadsides, where the peat is not cracked. Sloping sites will be more difficult to rewet because this method will only wet small parts of the area. Techniques can probably be developed to rewet sloping sites with cracked peat successfully, but their cost is liable to be high.

Chapter 6

Restoration results and practical experience

Evidence of successful restoration is limited and little has been published on the costs or results of attempts to restore bogs.

Evidence of success: water level rise

Very few reports of attempts to raise water levels by damming ditches have been published. There are even fewer relating to formerly afforested bogs. Anderson (1999) reported slight but significant raising of the water table resulting from bog restoration treatments used on pre-thicket and thicket stage forests adjoining pooled blanket bog in Caithness (Table 3). The combination of felling trees and damming plough furrows was more successful than either of these alone.

Installing a large number of dams does not guarantee successful rewetting. At Horse Hill Moss in Kielder Forest, over 700 dams were used on 1.5 ha of plough furrows but this left a confusion of hydraulic pathways, apparently unaffected by the damming (Clothier, 1995). Intensive experimentation would have been needed to get sufficient information on hydraulic pathways through the site for successful damming. Many of the dams appeared ineffective with only major drains being blocked successfully. New methods of retaining water on such sites are required.

A marked raising of the water table was achieved when bog restoration treatments were applied to an ombrotrophic bog in southern Finland which had been drained 28 years before (Komulainen *et al.*, 1998). Rewetting by removing the trees, blocking side drains and completely infilling main drains, raised the mean summer water table level from 31–36 cm below the surface before to 11–16 cm after.

Success was also achieved at Kirkkaanlamminneva, where summer water levels were raised considerably by using a digger to infill the ditches with the original spoil (Heikkilä and Lindholm, 1995c). However, a project at Koveronneva, in Central Finland, was unsuccessful. Damming drains only raised the water level by a few centimetres, insufficient to cause a visible vegetation change (Vasander *et al.*, 1992; Heikkilä and Lindholm, 1995c).

Because of the shortage of evidence from afforested bogs we must look to restoration projects on unplanted bogs. Wheeler and Shaw (1995) commented on the perceived success of peatland restoration projects at 43 sites in the British Isles, Holland and Germany. Ditches had been blocked or infilled at 27 of the sites. Rewetting was considered at least partly successful at almost all of these, indicators ranging from persistent flooding to a measured rise in water levels or, at some sites, the simple observation that the dams were holding water. At one site the ditch damming was inadequate to rewet the remnant mire, while at five others the rewetting was limited in extent and at a further five it was not completely successful. Most of the sites were partially cut away or completely cut over, so many needed the construction of bunds to hold water. In these cases, the bunds were usually successful in flooding the peat workings, but did not rewet the remaining primary surface effectively.

Evidence of success: vegetation succession

Many restoration projects have already been started. However, these have mostly been on

Table 3. Effect of blanket bog restoration treatments on water levels (Anderson, 1999)

Treatment	Depth to water table in summer (cm)	
	Wet month	**Dry month**
Do nothing (control)	**16**	**47**
Fell to waste *or* Fell and remove whole trees from site *or* Leave trees standing but dam plough furrows	**14**	**38**
Fell to waste and dam plough furrows *or* Fell and remove whole trees and dam furrows	**12**	**31**

drained or cutover bog. Few have been on afforested bogs and very few have published reports on vegetation responses to the restoration operations. This is partly because a lot of these projects are still in their early stages and partly because only a limited amount of vegetation monitoring has been done.

Restoration of Langlands Moss, a raised bog in Lanarkshire partially afforested in 1968, was started in 1995, when whole trees (trunks and branches, but not roots) were removed from the bog and drainage ditches dammed. The summer following these operations was unusually warm and dry and some of the *Sphagnum* which had remained under the trees perished following the sudden change in microclimatic conditions. One year later, *Sphagnum* cover had increased, with *S. cuspidatum* present in flooded ditches, *S. capillifolium* and *S. recurvum* in the damper microsites formed by the plough furrows, and *S. tenellum* colonising some areas of bare peat and of the conifer needle carpet which covered the previously forested ground (Brooks and Stoneman, 1997b). Some formal vegetation monitoring is taking place, the results of which may indicate successional changes in due course.

Results from Scandinavia have been mixed. Restoration began in 1987 at Koveronneva, an ombrotrophic mire in Seitseminen National Park, Central Finland. It had been drained and fertilized in 1970, but tree growth had still been poor. There was no visible vegetation response in the restored area and numerous tree seedlings were growing on the area where trees had been cleared, perhaps not surprising since the water table had hardly risen (Heikkilä and Lindholm, 1995c). Thus, early indications were that the restoration of Koveronneva was probably unsuccessful.

At Kirkkaanlamminneva (Heikkilä and Lindholm, 1995c) the trees were felled and their stems removed from the mire in 1992. Summer water levels rose and visible vegetation succession began within 2 years. Lichens (*Cladina* spp. and *Cladonia* spp.) which had colonised *Sphagnum* lawns after drainage, disappeared quickly after restoration. Hummock vegetation, which had been little altered by drainage, appeared not to respond to restoration. Part of the mire which, prior to drainage, had had a taller and thicker tree stand and a ground flora dominated by dwarf shrubs, still had a low water level after tree removal and forest species, such as *Vaccinium vitis-idaea* and *Trientalis europaea*, appeared to have benefited.

At Viheriäisenneva, an ombrotrophic bog in southern Finland, tree clearance and rewetting, which successfully raised the water table, caused increases in *Sphagnum balticum* and *S. fuscum* and decreases in typical forest mosses within two years (Komulainen *et al.*, 1998).

Published reports of vegetation change following bog restoration are difficult to interpret because the projects reported on are generally too recent for the direction of succession to be clear. There are some signs that bogs can be restored but there has been, as yet, no clear demonstration of success.

Costs of restoration operations

Costs have been obtained for case studies and are detailed in Appendix 2. They are used in this section to give examples of costs for each operation and to illustrate the factors affecting costs.

Tree removal

When trees are being cleared for the purpose of bog restoration it may be deemed necessary to remove them earlier than commercial forestry considerations would dictate. The net cost varies with timber volumes and value, accessibility, and how much material can acceptably be left on the site.

Costs for removing all material from the bog can be very high. The net cost of whole tree removal by helicopter from Langlands Moss, a Lanarkshire raised bog, ranged from £4300 to over £9000 per ha depending on timber volume and value (see Appendix 2).

Costs can be reduced by leaving valueless material on site, but the ecological consequences of doing so need to be considered (see Chapter 4). The net cost of completely removing small areas of pre-thicket and thicket stage forest to restore blanket bog in Sutherland (Wilkie *et al.*, 1997) and Caithness (Anderson, 1999; Wilkie, personal communication) was £1250 per ha. At the same sites the net cost was only £250 per ha if the trees were felled to waste (see Appendix 2). The suitability of young trees for the Christmas tree market can affect their value.

Costs can also be reduced by delaying restoration until the trees are more valuable. At age 25, estimated net costs of removing forest from East Flanders Moss, Stirling, ranged from £1100 to £2800 per ha, depending on whether the trees were felled to waste or harvested (see Appendix 2). If delayed until age 42, the estimated net cost was nil. The risk of windthrow has to be taken into account when considering delaying felling. A trade-off exists between the desirability of reducing costs and the undesirability of allowing peat cracking to reach a more advanced stage, affecting the feasibility and/or costs of rewetting the site.

Damming drains and furrows

The costs of damming drains or furrows depends on the size of dams needed and the intensity and average gradient of the drain or furrow network. The layout of the drainage system sometimes permits large areas to be rewetted by damming drains at a few key points.

Burlton (1996) gave costs, including material and labour, of installing four types of dam on the Border Mires (see Appendix 2). They range from £8 for a small plywood dam suitable for a narrow plough furrow to £132 or more for a plastic piling dam in a large drain. Wilson (1997) gave a cost of £190 for plastic piling dams in large ditches.

The intensity and cost of damming will be low if the layout lends itself to rewetting by blocking drains at a few key points. If not, a systematic approach will be needed. Brooks and Stoneman (1997a) gave a rewetting specification, in terms of the maximum water level drop at each dam, of 10–20 cm. This implies a damming intensity of 87–175 dams per degree average gradient per km of drain. The cost of damming drains at Langlands Moss was £1000 per ha.

A furrow damming intensity of 218–436 dams per degree average gradient per ha would be required on 4 m spaced double mouldboard ploughing (see Appendix 2). Single throw ploughing at 2.4 m spacing would require 363–727 dams per degree average gradient per ha.

In practice, it is often just the outer part of the bog that has been ploughed and afforested, with few drains on the central part. This is the situation on many of the Border Mires, where it is estimated that it will cost £50 per ha of mire (including the central areas which were not afforested) to get an average damming intensity of one dam per 20 m of drain or furrow.

Monitoring

At least until the feasibility of successful restoration is demonstrated, monitoring will be an essential part of restoration management. It is impossible to define a standard monitoring scheme because it needs to be tailored for the site and linked to the objectives which will differ between projects. In many cases water level objectives are likely, while in others vegetation changes will be aimed for. Monitoring techniques were described by Brooks and Stoneman (1997a).

Hydrological monitoring of four of the Border Mires, involving monthly readings of 4–6 water level range gauges at each, was estimated to cost £990 per year (at 1999 levels) altogether (Burlton, 1996). Equipment and installation costs were not included but these are estimated to have been around £1700. Monitoring the vegetation of the same four bogs using permanent quadrats cost £900 per year plus a one-off installation cost of around £550. If monitoring was continued for 10 years, the combined cost of monitoring both water levels and vegetation amounts to £520 per year per site.

Assessing the cost-effectiveness of restoration

It is very difficult to judge the cost-effectiveness of bog restoration projects because the desired and actual end-points are often unclear, the cost of restoration may be inextricably tied up with that of the land-use change involved, and the benefits cannot be expressed in monetary terms.

In 1996 Scottish Natural Heritage paid £1.8 million to buy out peat harvesting rights on a small part of East Flanders Moss. That sum could be regarded as contributing towards conserving the whole bog, on the grounds that water level would be lowered over the entire area as a result of harvesting peat from a relatively small area. In the same way, the restoration of the 40 ha plantation might be regarded as crucial for the conservation of the whole bog. The validity of spreading the costs in this way needs to be determined.

Factors expected to influence the cost-effectiveness of restoration attempts are listed in Table 4.

Table 4. Factors influencing the feasibility and cost-effectiveness of restoration

Aiding cost-effective restoration	Hindering cost-effective restoration
Flat or with very shallow gradients (makes rewetting easier)	Sloping site (makes rewetting more difficult)
Presence of key points in drainage system where blocking drains would rewet large areas (makes rewetting much cheaper)	Absence of key points in drainage system (makes rewetting far more expensive)
If no scope for rewetting by damming key points in drainage system, peat not cracked between plough furrows (scope for rewetting by damming plough furrows)	Peat severely cracked between plough furrows (makes rewetting by damming plough furrows ineffective)
Intact, not cutaway, bog edges (no fundamental constraint to keeping the bog wet)	Cutaway bog edges (continue to exert a drying influence on the adjacent bog)
Maximum distance to a hard road <1 km (may allow harvesting without road-making)	Maximum distance to a hard road >1 km (may necessitate road-making)
Pure Sitka spruce or SS/LP mixed forest (sufficient quantity and quality of brash for making brash mats during harvesting)	Pure lodgepole pine forest (possible harvesting problems due to brash shortage and rapid breakdown)
First rotation forest canopy not yet closed (presence of remnant bog vegetation)	Forest canopy closed (little or no remnant bog vegetation)
Wide rides (presence of remnant bog vegetation as a source for recolonisation)	Narrow rides (little or no bog vegetation to act as a recolonisation source)
Unplanted bog in forest (presence of remnant bog vegetation as a source for recolonisation)	No unplanted bog in forest (no bog vegetation to act as a recolonisation source)
Adjacent to unplanted bog (presence of bog vegetation as a source for recolonisation)	No adjacent unplanted bog (little or no bog vegetation to act as a recolonisation source)

Chapter 7

Conclusions

- The effectiveness of bog restoration projects cannot yet be assessed because it must be judged over a term which allows the direction of succession to become clear (10 years minimum) and all the post-forestry projects so far undertaken are too recent.
- There is not enough evidence that post-forestry bog restoration can succeed to justify undertaking bog restoration on a large scale.
- There are limited early signs of success. These justify doing further research and encouraging a limited number of new projects. They should be carefully monitored to provide information for reviewing bog restoration policy in future.
- It is necessary to raise the water table to rewet sites sufficiently to restore bogs.
- Costs of rewetting sites using current techniques (i.e. manually installing sheet or board dams in drains and/or plough furrows) are highly variable and depend on site-specific factors, such as ground slope and the degree to which water levels are controlled by discrete blockable drains.
- There is scope for developing cheaper methods of rewetting bogs.
- The main gaps in our knowledge are: a) the feasibility of recreating high quality bog habitats on afforested bogs, b) the thresholds, in terms of the water level regime, for successful bog restoration, c) the influence of bog restoration operations on nutrient availability and the influence of nutrient release on restoration success, and d) the potential for increasing the cost-effectiveness of bog restoration operations by technical development. Further research is needed to close these gaps.

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Appendix 1

Forestry Commission policy on bog restoration

Extract from Forestry Commission Guideline Note 1 *Forests and peatland habitats* (Patterson and Anderson, 2000)

Conserving and restoring bogs in existing forests

The Forestry Commission will encourage the conservation of peatland habitats within forests as part of the design and management of open ground, which is normally expected to form 10–20 % of the total area of woodland. Restoration of former bog habitats may be possible within some of the larger openings in extensive forests. The creation of transition zones at planted forest edges adjacent to open blanket bog will also be encouraged. The possibility of developing new forms of wooded bog will be researched, especially those which may contribute to the wet woodland Habitat Action Plan and expansion of the European Union priority habitat type Bog Woodland. The Forestry Commission may offer grant-aid towards the cost of bog restoration operations such as drain blocking or the removal of unwanted natural regeneration in areas forming part of the open-ground component of woodlands. Forest Enterprise will carry out similar work in Forestry Commission forests where it is considered appropriate to do so.

There is a lack of firm evidence on the prospects of successfully restoring raised or blanket bog ecosystems after woodland removal. Further research is required to establish the costs and benefits of bog restoration on a variety of scales and conditions. The Forestry Commission is carrying out research of this type, and will continue to do so in partnership with others.

In the meantime (because the Government's policy is to maintain and expand woodland cover), bog restoration projects on a scale which exceeds the normal open ground provision within woodlands will only be approved by the Forestry Commission where there are high net environmental benefits to be obtained from permanent tree removal. An Environmental Impact Assessment may be required by the Forestry Commission to help reach a decision.

In such cases the Forestry Commission may give felling approval without a replanting condition and may also decide not to pay grants for restocking an area either by planting or by natural regeneration. Similarly, Forest Enterprise may decide not to replant. Such special cases in both private and public forests are likely to be found on deep peat sites (average depth 1 m or more), which are judged to have a high probability of successful restoration to active raised bog or active blanket bog; and are either hydrologically linked to significant remnant areas of active bog or are adjacent to them and important to their ecological integrity. Important archaeological or landscape benefits may sometimes justify felling without subsequent replanting in some peatland areas, especially when combined with ecological benefits.

A key requirement for Forestry Commission support for bog restoration projects will be an agreed management plan which sets out how restoration is to be achieved and who will carry out the work over an adequate timescale. The Forestry Commission will seek to work with partner organisations to develop management plans.

Appendix 2
Examples of the cost of restoration operations

Table A1. Costs of whole tree removal by helicopter at Langlands Moss, Lanarkshire (Brooks and Stoneman, 1997; Wilson, 1997). The forest was 27 years old but about half consisted of worthless, spindly trees which had self-seeded following a fire in 1979

Operation	*£/tonne	*£/ha
Chainsaw felling	13	800
Roping up bundles	15	1000
Helicopter extraction	72	4700
Off-bog delimbing and shredding	45	3000
Management and road repairs	19	1300
TOTAL	163	10 800
Revenue from 330 tonnes timber	-28	-1800
TOTAL NET COST	136	9000

* Costs expressed at 1999 levels and based on 5 ha of forest. Weight basis is saleable timber only.

Table A2. Net costs of experimental treatments for clearing (a) pre-thicket stage forest in Sutherland (Wilkie *et al.*, 1997) and (b) pre-thicket and thicket stage forest in Caithness (Anderson, 1999; Wilkie, personal communication). The Sutherland site had some trees suitable for the christmas tree market

Site	Treatment		Net cost (*£/ha)
(a) Inchkinloch (10-year-old forest)	Fell to waste, leave whole trees lying		250
	Fell to waste, sned		410
	Fell to waste, sned, put debris into furrows		520
	Fell to waste, windrow in every 6th to 8th furrow		580
	Fell and remove whole trees from site		1250
(b) Halsary (11 years old) and Braehour (16 years old)	Fell to waste, leave whole trees lying	pre-thicket	630
		thicket	630
	Fell to waste, sned	thicket	740
	Fell and remove whole trees from site	pre-thicket	1250

* Costs expressed at 1999 levels

Table A3. Estimated costs of tree removal options at East Flanders Moss. Half the 25 year old forest was badly checked Sitka spruce and half was well grown lodgepole pine

		*Cost (£/ha)
Fell now:	fell to waste, roughly sned, cut into 1 m lengths	1100
	whole tree extraction by skyline	2800
	fell spruce to waste, harvest pine timber only	1600
Fell spruce to waste, delay harvesting pine until age 35		1000
Delay harvesting until age 55		-1600

* Costs expressed at 1999 levels

Table A4. Costs of damming drains and plough furrows on the Border Mires in Kielder Forest (Burlton, 1996)

Dam type	Size (m)	*Materials (£ per dam)	*Labour (£ per dam)	*Total (£ per dam)
Small plywood	0.6 x 0.9–1.2	6	2	8
Medium plywood	1.2 x 0.9–1.2	11	6	17
Large plywood	0.6–1.2 x 1.8–2.4	22	11	33
Large plastic	larger	110	22	132

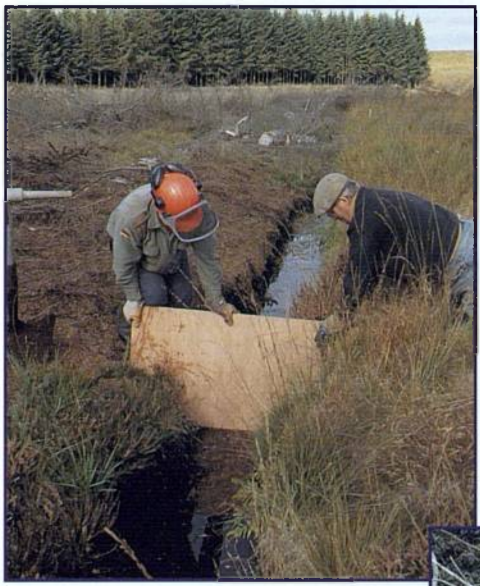
* Costs expressed at 1999 levels

Table A5. Damming intensity and cost of damming 4-m-spaced plough furrows on a range of gradients using two different rewetting specifications. Costs based on £17 per dam

Furrow gradient	10 cm height between dams		15 cm height between dams	
	dams/ha	*£k/ha	dams/ha	*£k/ha
0.5°	218	4	145	2
1°	436	7	291	5
1.5°	654	11	436	7
2°	872	15	582	10
3°	1308	22	872	15
4°	3583	61	1163	20

* Costs expressed at 1999 levels

Peat bogs have been used for forestry because they are not well suited for agriculture. The role of peatlands as habitats for specialist flora and fauna is now widely recognised and nature conservation has become a major factor in decisions about their use. Forestry Commission Guideline Note 1 *Forests and peatland habitats* sets out current policy on this subject.



On afforested peat bogs, forestry and conservation objectives often conflict. Where the two interests cannot be reconciled and the conservation objectives take precedence, deforestation and restoration to bog is desirable. But what does this entail? Is it feasible? Will it re-create worthwhile peatland habitats? And how much does it cost?

This Technical Paper attempts to answer these questions by reviewing relevant scientific and 'grey' literature and unpublished material. It draws on lessons learned in restoring bogs damaged by other land uses.

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