Ten-year results of a comparison of methods for restoring afforested blanket bog

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SUMMARY

There is growing interest in the restoration of blanket bogs that were afforested during the 1960s to 1980s, to avoid further loss of carbon to the atmosphere and to regain and defragment important blanket bog habitat. This paper reports the findings from a ten-year experiment in the UK to test the effectiveness of restoration treatments on water table depth, peat bulk density and water content, ground surface height and vegetation development. Treatments used were the six combinations of damming or not damming plough furrows with leaving the trees alive, felling and leaving them on the ground or felling and removing them. Combining felling with damming furrows was most successful in raising the water table, whether or not the felled trees were removed. Only where felling was combined with damming did the water table continue to recover between Years 5 and 10. Over ten years, the water level in these treatments rose to slightly below that of non-afforested reference bog at the same sites. This occurred as a rapid initial rise, following which there was only very slight further improvement. Felling caused the species composition of the vegetation to change towards that of the reference bogs. The process was slow, with the vegetation becoming more dissimilar to non-forested reference bog in the first five years and then becoming more similar to the reference bog after Year 5. Surprisingly, damming plough furrows had little effect on the vegetation except that, in combination with felling, it increased differentiation between the plough furrows and other positions on the ploughed ground. Conifer seedlings established on the restored plots, most densely where they adjoined standing forest, and had similar density and growth in all the felled treatments. The restoration treatments resulted in a decrease in bulk density and increase in water content of the upper peat, probably due to an unloading effect caused by the raised water table buoying up the drained peat layer. In some treatments this was amplified by removal of the weight of the trees. Damming the plough furrows caused a 5–7 cm rise in ground surface height, suggesting that subsidence resulting from primary consolidation and secondary compression is at least partly reversible.

KEY WORDS: forestry, mire, restoration, restore, subsidence, uplift, vegetation, water table

INTRODUCTION

Peatlands are internationally recognised as important for the maintenance of global biodiversity and for carbon storage vital to the world's climate system (Ramsar Convention Contracting Parties 2002). Restoration of damaged and degraded peatlands is Priority Project 1 of 12 in the Scottish Government's biodiversity plan (Scottish Government 2015). Blanket bog is the predominant habitat type within blanket mires, which are extensive peatlands that can occupy whole landscapes. Blanket bog landscapes are found in areas with a cool, wet climate where rainfall exceeds evaporation for most of the year so that peat accumulation occurs not only in basins but also on flat and gently to moderately sloping ground. On a world scale, blanket mires are largely restricted to areas with a hyperoceanic climate in a few high latitude regions of both hemispheres (*i.e.* the British Isles, Iceland, Norway, Newfoundland, Alaska, Kamchatka, Japan, Tierra del Fuego, the Falkland Islands, Tasmania and New Zealand) (Lindsay 1995). Britain is estimated to have approximately two million ha of blanket peat, which is thought to be 10–15 % of the world's total area (Lindsay 1995). Blanket bog is a priority habitat type for conservation in the European Union and was the subject of a Habitat Action Plan within the UK Biodiversity Action Plan (UK Biodiversity Steering Group 1995, UK Biodiversity Group 1999). Scotland's National Peatland Plan recognises blanket bog as Scotland's largest terrestrial carbon store at 1.6 billion tonnes of carbon and places safeguarding this and enhancing its capacity to store further carbon among its highest priorities (Scottish Natural Heritage 2015).

Conversion of peatland to forestry through afforestation (*i.e.* preparing and planting open bogs) or forestry drainage (*i.e*. drainage of natural sparse or slow-growing peat woodland) has been a significant cause, and in some countries the major cause, of peatland habitat loss in northern Europe and parts of North America. For example, in Britain and Ireland, techniques for industrial-scale afforestation of peatland were developed in the 1960s and many blanket bogs were afforested during the 1970s and 1980s. While in most of Europe and North America peatland forestry employs forest drainage to encourage development and more vigorous growth of a largely pre-existing tree cover, UK peatland afforestation practice planted treeless open bog. Establishment of a forest involved spaced furrow ploughing with a single or double mouldboard plough at 2 m or 4 m intervals respectively (Zehetmayr 1954, Mason 1999), and digging or ploughing deeper drains in strategic places to lead surface water from the plough furrows off the site. Plough furrows are open channels 0.3–0.45 m deep and 0.4–0.9 m wide, which lower the water table and provide the material for a ridge of bare peat thrown up on one or both sides on which to plant nursery-grown trees. Forestry drains are larger open ditches, initially 0.9 m deep and 1.0–1.5 m wide, that collect water from the furrows and channel it to a stream through a riparian buffer zone. Tax changes in 1988 removed the incentive for further investment-driven afforestation of UK peatlands and peatland afforestation activity ceased by the early 1990s (Warren 2000). By 2000, UK government policy regarding forestry on peatland included a presumption against further afforestation on deep peat and supported restoration of the most valuable sites (Patterson & Anderson 2000). By 2014, the Scottish Government was funding a substantial programme of peatland restoration. In the case of afforested peatlands, a strategic approach that balances the biodiversity and carbon benefits of restoration against the loss of productive woodland and potential associated carbon gain was adopted (Forestry Commission Scotland 2014, Forestry Commission Scotland 2015).

Early restoration initiatives, such as the 1989 Border Mires restoration programme at Kielder in northern England (Lunn & Burlton 2010) highlighted the lack of information on best practice. Consequently, it was recognised that there was a growing need for research to assess the feasibility and effectiveness of different restoration methods in dealing with the changes resulting from afforestation. More recently, concerns have emerged that blanket bog habitat on protected sites adjoining forests could be adversely affected by forest-edge effects such as drainage and avoidance by valued breeding bird populations (J.D. Wilson *et al*. 2014). Possible impacts of afforestation that need to be reversed to restore blanket bog, including ground outside the forest affected by forest edge effects, are:

1. deep shading of the ground surface by the forest canopy;
2. lowering of the water table by the network of drains and plough furrows combined with increased evapotranspiration (Sarkkola *et al.* 2010, Sarkkola *et al.* 2013) by the tree layer;
3. change in composition of field and ground-layer vegetation; and
4. drying, shrinkage and wastage of the upper peat layer.

An additional consequence that may or may not need to be reversed is:

5. compaction and consolidation of peat below the upper layer.

In 2001, a literature review revealed that there were very few published experimental results or descriptions of methods for restoring afforested peatlands (Anderson 2001). Initial findings from restoration projects in Finland reported mixed success in raising water levels and restoring the pre-drainage vegetation communities of the sites (Vasander *et al*. 1992, Heikkila & Lindholm 1995a, 1995b; Komulainen *et al*. 1998). Descriptive accounts of some restoration projects in Britain suggested that techniques varied hugely in cost and that they could still fail (Clothier 1995, Brooks & Stoneman 1997a, Parkyn 1997, Wilkie *et al*. 1997, E. Wilson 1997a, 1997b). Since then, further results of experiments and trials on restoration of afforested or forestry-drained peatlands have been reviewed by Andersen *et al.* (2016). These various studies revealed, among other things, that felling the trees and damming the drains on an afforested raised bog raised the water table to near the surface and changed the vegetation towards its natural composition with little difference among different treatments except in the rate of vegetation recovery (Anderson 2010). The natural testate amoeba community, regarded as a good indicator of peatland wetness, recovered best where some tree remains were left on the ground after felling, rather than being completely removed, probably due to a mulching effect of the felling debris (Vickery & Charman 2004).

In the Scandinavian forestry drainage context, restoration of forestry-drained bog and fen peatlands by clear-cutting, slash removal and a combination of ditch damming with peat and complete infilling with the old spoil ridges caused the water table to rise rapidly and remain close to the peat surface (Komulainen *et al.* 1999, Jauhiainen *et al.* 2002, Haapalehto *et al.* 2011). The vegetation changed more rapidly at first on a fen peatland than on a bog (Komulainen *et al.* 1999), with forest species declining and the mire species *Eriophorum* *vaginatum* increasing substantially (Jauhiainen *et al.* 2002). Over ten years, the vegetation changes continued towards the target communities (*i.e.* those that existed on the respective sites prior to drainage) but many typical species had still not returned after a decade (Haapalehto *et al.* 2011). Mineral element concentrations indicated that natural nutrient cycling between peat and plants had returned within ten years. There were only minor differences in the recovery of the restored bog and fen sites, with Ca, K, Mg and Mn concentrations recovering to natural levels in the bog but only Ca approaching natural levels in the fen. Vegetation changed more rapidly in the fen than in the bog, with *Eriophorum vaginatum* increasing sharply and then declining in the fen but increasing more gradually over the ten years in the bog. All the fen-specific *Sphagnum* species failed to recolonise the fen within ten years whereas *Sphagnum balticum*, a wet hollow species of pristine bogs, made a recovery in the bog. The most recent results show signs of some key peatland ecosystem features and functions recovering within ten years or less of restoration (*i.e.* surface layer growth, *Sphagnum* cover and production, specialist invertebrate communities) (Kareksela *et al.* 2015, Maanavilja *et al.* 2015, Noreika *et al.* 2015), while other features failed to recover or recovered only partially (*i.e.* plant community composition, surface layer carbon sequestration) (Kareksela *et al.* 2015).

As for forest edge effects on neighbouring areas of blanket bog, Shotbolt *et al*. (1998) demonstrated subsidence of blanket peatland for up to 40 m outside forest plots at Bad a' Cheo over the 30 years since the forest was planted. Manzano (2012) reported a 50–100 m zone of self-seeded conifers on blanket bog adjoining forests in Strathmore. J.D. Wilson *et al.* (2014) reported a forest edge effect that deterred breeding by dunlin *Calidris alpina* (L.) and European golden plover *Pluvialis apricaria* (L.) on otherwise suitable habitat within 700 m of forest edges.

One key uncertainty associated with restoration after afforestation concerns the quantity and nature of tree material which should be removed after felling in order to promote restoration of the original peatland habitat type. In particular, it is unclear whether it is necessary to remove foliage and branches along with the saleable timber or, in cases where pre-commercial felling would be involved, whether merely felling the trees to waste (*i.e.* leaving them to rot on the ground) is sufficient. This may also affect the condition of the restored habitat. There are concerns that nutrient release from the felled trees, especially from the foliage, might enrich the peat and encourage the development of more eutrophic vegetation than the equivalent pristine peatland communities (Anderson *et al*. 1995). Until now, conservation guidance on dealing with trees on bogs has taken a precautionary approach by advocating removal of cut material (Brooks & Stoneman 1997b, Dupieux 1998, Foss & O'Connell 1998, Brooks *et al*. 2014). Another area of uncertainty is the question of whether it is necessary to dam the drains and/or plough furrows. In addition, it seems self-evident that in order to restore peatland vegetation, the ground needs to be wetter than it is under forest, but it is unclear whether the reduction in evapotranspiration achieved by felling trees alone will be sufficient. It is not known to what extent natural blockage of the drains and furrows by *Sphagnum* growth after felling will raise the water table. If not sufficiently, then restoration may require damming of the drains and plough furrows. A further question that arises is whether it is necessary to fell the trees at all if drains and furrows are being dammed, as they might die due to root waterlogging, as observed by Meade (1992) following re-wetting of a birch-covered cut-over raised bog. Lastly, the extent to which peatland restoration is capable of reversing forest edge effects on adjacent blanket bog is unknown.

Research aim

To explore at least some of these issues in relation to the restoration of afforested blanket bog, an experiment was set up in Scotland in 1996 to determine the effectiveness of different combinations of felling and furrow-damming treatments in restoring ground conditions suitable for renewed bog growth. Some early results of this experiment have already been reported (Anderson 2010). In this paper we present the results of the experiment through the period of ten years after felling the forest. In particular, we evaluate the effectiveness of tree felling, tree removal, and drain and furrow blocking as restoration techniques for afforested blanket bog. We report results for depth to water table, species composition of vegetation and ground surface height, as well as peat water content and bulk density, as measures indicating the degree of success of the treatments in reversing the impacts of afforestation discussed above.

METHODS

Experimental sites and design

The experiment used three afforested blanket bog sites in Halsary and Braehour Forests in Caithness, northern Scotland, UK (Figure 1). Site details are given in Table 1. A randomised block design was used. The experiment was established between

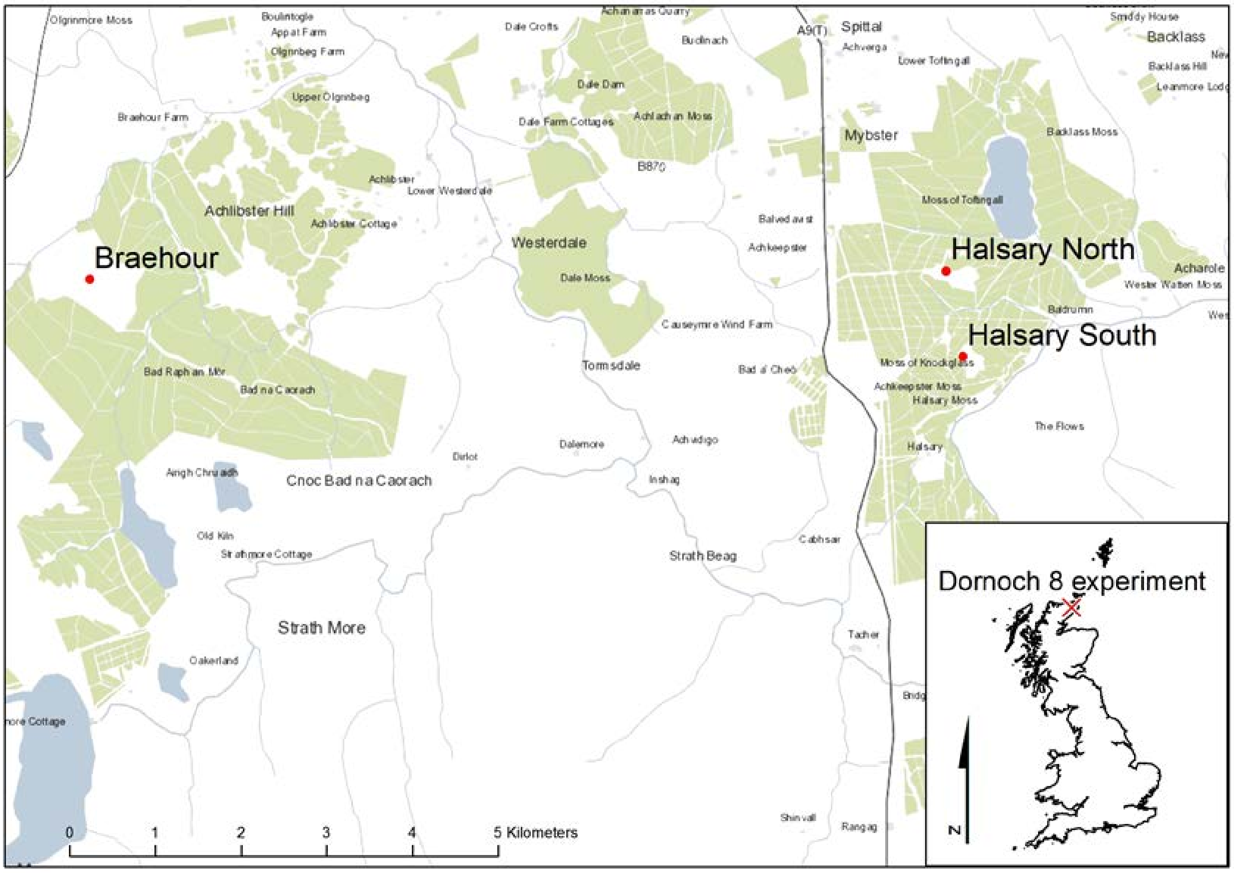

Figure 1. Locations of the experiment (red cross on inset map) and its three sites (red dots on main map).

October and December 1996. Two blocks of treatment plots were set up at each of the three sites, giving a total of six replicate blocks. The two blocks at each site were 600 m × 40 m strips of forest edge lying on opposite sides of an unplanted area of bog and each block was divided equally into six mostly contiguous plots (Figure 2). The plots were not fenced so were open to deer present in the forest.

The six treatments applied within each block were the six possible combinations of two damming treatments with three felling treatments (Table 2). The damming treatments were - O: not damming plough furrows or drains at all and, D: damming plough furrows, and any drains, at 20 m intervals. Each dam was made from 3 or 4 interlocked 0.3 m wide plastic piles, either 0.8 m or 1.2 m deep, depending on forest age and peat depth. The felling treatments were - O: not felling the trees, F: felling the trees and leaving them lying intact where they fell, and R: felling the trees and either removing them whole from the felled area or debranching them and cutting them up so they lay flat on the ground. This last treatment had to include the two alternative sub-treatments because the trees in Blocks 5 and 6 were too big to be dragged off the treatment plots manually. Debranching and cutting them up was considered the best practical alternative to actually removing them. This was taken into account in the statistical analysis and interpretation.

Measurements

The quantities measured or estimated, the methods used and the frequency of measurements are summarised in Table 3. As well as giving a comparison of treatments *versus* control, the monitoring and assessment programme was also designed to provide some intermediate points on the ten-year time series and to see how the treatments influenced edge effects at the boundaries between the remaining forest and the restored bog and between the restored bog and the adjacent non-afforested bog.

Rainfall

Rainfall was measured using rain gauges on open bog adjoining each of the experiment sites. Six rain gauges, one *per* block, were emptied and recorded monthly. This made it possible to relate water-table behaviour to rainfall inputs.

Table 1. Experiment site details.

	Blocks 1–2 Site 1	Blocks 3–4 Site 2	Blocks 5–6 Site 3
Site name	Halsary South	Halsary North	Braehour
Latitude	58° 25.8′ N	58° 26.3′ N	58° 26.3′ N
Longitude	3° 23.5′ W	3° 23.7′ W	3° 34′ W
UK National Grid Reference	ND186503	ND183513	ND083512
Altitude (m above sea level)	85	85	95
Mean annual rainfall (mm) ‡	1000	1000	1000
Mean days rain $\geq$ 1 mm‡	170	170	170
Peatland type	blanket bog	blanket bog	blanket bog
National Vegetation Classification type §	M18	M18	M18 + M19
Maximum slope (degrees)	0.4	0.4	1.5
Peat depth (m)	3.3–6.0	5.5–6.7	0.8–6.8
Peat dry bulk density on adjacent bog (Mg m^{-3})			
10–20 cm depth	0.07	0.08	0.10
70–80 cm depth	0.07	0.06	0.08
Peat dry bulk density in forest (Mg m^{-3})			
10–20 cm depth	0.10	0.10	0.13
70–80 cm depth	0.08	0.07	0.11
Forest tree species †	LP/SS mix	LP/SS mix	LP
Forest canopy closure stage	closing	closing	closed
Forest age when treatments were applied (yr)	11	11	15
Forest stocking density (stems ha^{-1})	2775	2127	2485
Mean tree diameter at 1.3 m (cm)	5.3	6.5	11.5
Mean tree height (m)	3.2	3.7	6.5
Plough furrow spacing (m)	4.0	4.0	4.0
Peat aeration depth (m)*	0.2–0.3	0.2–0.3	0.4–0.6

‡ Source: www.metoffice.gov.uk/climate/uk/averages
† LP = lodgepole pine (*Pinus contorta* Douglas ex Loudon), SS = Sitka spruce (*Picea sitchensis* (Bong.) Carr.)
* Peat aeration depth estimated by observing the depth of darker peat in 0.9 m long cores
§ National Vegetation Classification (Rodwell 1991). M18 = *Erica tetralix-Sphagnum papillosum* raised and blanket mire. M19 = *Calluna vulgaris-Eriophorum vaginatum* blanket mire.

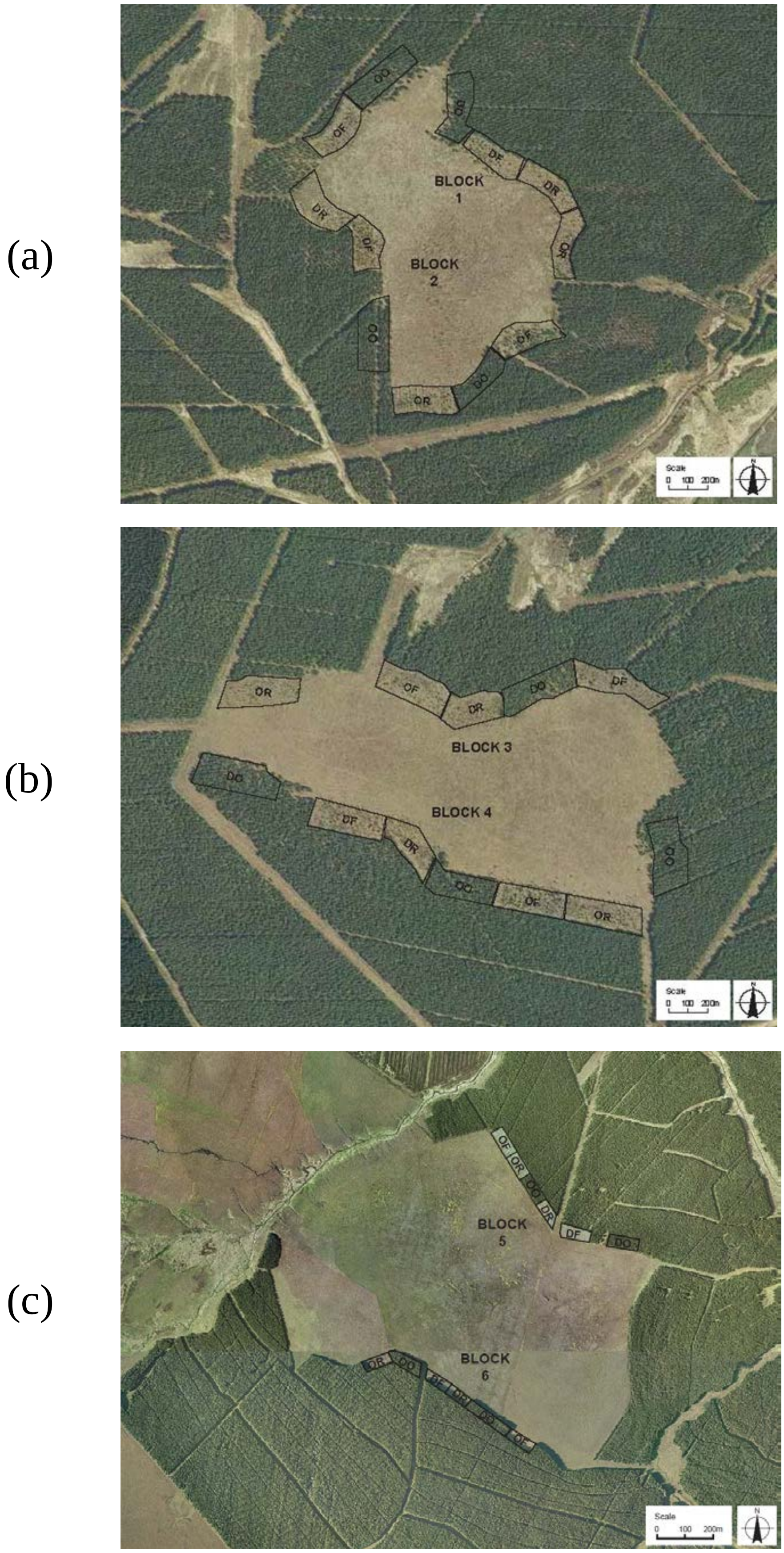

Figure 2. Aerial photographs showing the layout of blocks and plots at (a) Halsary South, (b) Halsary North and (c) Braehour.

Table 2. Experimental treatments.

		Damming treatments	
		O (no dams)	D (dams)
Felling treatments	O (no felling)	**OO** (control): no dams installed and forest left to continue growing	**DO**: plough furrows and drains dammed but trees left growing
	F (trees felled to waste)	**OF**: no dams installed; trees felled and left intact lying on bog surface	**DF**: plough furrows and drains dammed; trees felled and left intact lying on bog surface
	R (trees removed)	**OR**: no dams installed; trees removed, *i.e.* felled and either dragged whole off the site (Blocks 1–4) or de-branched and cut up to lie flat on bog surface (Blocks 5–6)	**DR**: plough furrows and drains dammed; trees removed, *i.e.* felled and either dragged whole off the site (Blocks 1–4) or de-branched and cut up to lie flat on bog surface (Blocks 5–6)

Table 3. Summary of measurements.

Response variable	Variable measured	Method	Number	Schedule
water table	rainfall	storage rain gauge	1 *per* block	monthly Apr–Oct Years 1, 2, 5, 10
	water table depth	dipwells	16 *per* plot	monthly Apr–Nov Years 1, 2, 5, 10
		WaLRaGs	2 *per* plot	monthly Apr–Nov Years 1, 2, 5, 10
	aeration depth of peat	depth of dark peat in cores	8 *per* plot	once Years 0, 5
vegetation	vegetation of treated areas	1 m^2 fixed quadrats	8 *per* plot	once in summer Years 0, 2, 5, 10
	vegetation of adjacent bog	1 m^2 fixed quadrats	3 *per* plot	once in summer Years 0, 5, 10
	tree seedling density	40 m × 2 m transects	2 *per* plot	once in winter Years 0–10
ground surface height and peat properties	ground surface height	surveyor's automatic level	51 transect points plus 1 benchmark *per* plot	once in autumn Years 0, 6, 10
	water content of peat	weigh core sections fresh and oven-dry	8 *per* plot × 2 depths	once in autumn Years 0, 6, 11
	wet and dry bulk density of peat	weigh core sections fresh and oven-dry	8 *per* plot × 2 depths	once in autumn Years 0, 6, 11

Water table depth

Dipwells and water level range gauges (walrags) (Bragg *et al*. 1994) were used to measure water table depth from the original ground surface beside the plough ridge. The dipwells were 0.9 m deep holes made with a 5 cm × 5 cm square peat corer and lined with perforated PVC pipe, 5 cm in diameter, protruding 10 cm above the surface. The pipes were anchored to the surface peat using pins of galvanised fencing wire and capped with plastic end-caps. The dipwells were read using an audio dipstick designed at Forest Research. The walrags consisted of a float with a pointer attached inside a perforated borehole so that float movement pushed foam blocks up and down within a calibrated vertical channel by which they indicated the maximum and minimum levels between dipwell readings. The walrags were anchored to the ground surface by bolting a horizontal 33 cm length of 3 cm diameter plastic waste pipe to the outside at ground level and pinning this to the ground.

Dipwells were arranged at 10 m intervals along two transects, 10 m either side of the plot centre, extending out onto the adjoining bog in one direction and into the adjoining standing forest in the other, so that minimum edge effects at least could be determined. Eight of the 16 dipwells were located within the treatment plot. One walrag was sited in the centre of each plot and a second located 10 m from the edge of the plot on the adjoining bog. In each block, an additional walrag was installed on open bog at least 50 m away from the forest edge to act as a non-afforested reference plot. Dipwells and walrags were read monthly from April/May to October/November during the first, second, fifth and tenth years after the treatments were applied.

Vegetation

Permanent quadrats were used to monitor the composition of vegetation. These were arranged contiguously on short belt transects running perpendicular to the ploughing in two places, totalling 8 m^2 *per* plot. The individual quadrats represented a plough furrow (two quadrats), a plough ridge (two quadrats each consisting of two separate 0.5 × 1.0 m half quadrats combined) or the original surface between these (four quadrats). The percentage cover for species or species groups was estimated by eye to the nearest 1 %. Another three quadrats were located on the open bog adjoining each plot, at distances of 5 m, 15 m and 25 m from the original forest edge. The data from these were used as a reference to determine whether the vegetation of the treatment plots was reverting towards a similar composition. It is acknowledged that this open bog vegetation could have changed due to forest edge effects during the period in which a forest grew beside it. We think that any change during the eleven years (Blocks 1–4) or 15 years (Blocks 5–6) of adjacent forest growth would have been slight and that the open bog quadrats are still a useful reference, albeit not a pristine one.

Tree regeneration was assessed annually using two 40 m × 2 m belt transects running across the plot, perpendicular to the forest edge. Seedling species and heights were recorded.

Ground surface height

Ground surface height inside the former forest edge and on the adjoining bog was measured in November 1996, June 2002 and December 2006 to determine whether the treatments prevented further subsidence or even caused uplift of the ground surface. One transect was used for each plot, starting 10 m inside the treatment areas and running across the forest edge and 40 m out over the adjoining bog. Ground surface transects could not extend over the whole of the treatment area because it was impossible to survey more than 10 m through the trees for the baseline measurement and for subsequent measurements in the unfelled treatments. The ends were marked with posts, and a tape was temporarily stretched between the posts to locate the measurement points at 1 m intervals. Ground surface height was measured relative to a fixed benchmark using a surveyor's graduated staff and automatic level. The benchmark was the top of a sectional metal rod hammered firmly into the clay beneath the peat. The base of the staff was nestled down through the field layer vegetation so that it rested under its own weight on the bottom layer vegetation or the ground surface. To ensure consistency, the same person held the staff for all 36 transects during an individual surface height survey. However, this was a different person on each of the three measuring occasions.

Peat properties

Peat water content was measured for two depth layers, 10–20 cm and 70–80 cm. These layers were chosen to give a wide range of depth within the pre-afforestation peat profile and a lower layer beneath the water table. We avoided sampling the 0–10 cm layer because it contained conifer litter deposited above the pre-forestry peat surface. We also avoided the 80–90 cm layer because it was occasionally damaged by the box corers and it was convenient not to have to re-take the core when this occurred. Cores were taken with a square or box corer (Cuttle & Malcolm 1979), from the dipwell holes before treatment and from near these locations six and eleven years after treatment. A box corer prevents undue compression of the core by initially cutting three sides while the fourth continues to support the core. When the fourth side is cut, the peat core is supported by the three sides in contact with the corer. Water content was determined by weighing the cores before and after drying to constant weight at 105 °C. To detect whether the peat expanded in response to the restoration treatments, fresh and dry core weights were combined with measured corer volumes to calculate wet and dry bulk densities.

Statistical methods

Where multiple measurements were made within plots, statistical analysis was performed on plot means. Analysis of variance (ANOVA) and analysis of covariance (ANCOVA) were generally used to test the significance of treatment differences. ANCOVA was used where pre-treatment measurements provided the covariate. ANOVA and ANCOVA were applied separately for each of the four repeated post-treatment observations. Minitab Release 13 (Minitab Inc. 2000) was used to perform these analyses.

In spite of the inconsistency in the OR and DR treatment specification amongst blocks (see Table 2), the experiment was analysed as a single entity because the other treatments were consistent over the six blocks. The results were interpreted taking these differences into account. A second set of covariance analyses on plant species cover was performed using the data from Blocks 1–4 only, in order to look for any effects of removing felled trees from the site.

To remove the variability in the water table data resulting from seasonal and rainfall-related water table fluctuations, each year's mean depth to water table was expressed as a 'recovery index' (RI1) by scaling it between 0 and 1 according to its position relative to the control (OO) (RI1=0) and the unplanted open-bog reference (RI1=1). A second water table recovery index (RI2) was used to scale each year's mean water level according to its position relative to the Year 1 level for the control and each individual year's mean level for the unplanted open bog. RI2 was the same as RI1 for the first year after restoration but differed thereafter. Unlike RI1, RI2 is not immune to the effect of between-year variations in rainfall.

Plant community change was investigated using correspondence analysis, a multivariate ordination method. Species that occurred occasionally (*i.e.* in < 5 % of quadrats) were down-weighted, increasingly so the more infrequent their occurrence, because they represent noise in the data and could confuse its interpretation (*e.g*. Kent & Coker 1992). Quadrats on open bog adjoining the plots were used as a reference representing open bog vegetation, to determine whether the communities of the treated areas were reverting towards a similar composition. All the bog quadrats were used, regardless of whether they were 5 m, 15 m or 25 m away from the former forest edge because ordination showed no difference in the loci or centroids for the three distances. An ordination of the species composition data for all quadrats is used to illustrate and interpret the degree of recovery in the various treatments ten years after they were applied. Separate correspondence analysis ordinations for the three ploughing positions (ridge, original surface and furrow) are used to illustrate plant community trajectories (Matthews & Spyreas 2010) over the ten years. To avoid undue complexity we restricted this exercise to the first two ordination axes and interpreted a change that was more towards than away from the reference as progression, unlike the example in Figure 1c of Matthews & Spyreas (2010) where such a change is interpreted as a deviation. The Sorensen Index (*e.g.* Kent & Coker 1992) was used to express similarity between quadrats. This index is calculated for any two quadrats 'Q1' and 'Q2' by:

$$\text{Sorensen Index} = 2a / (2a+b+c) \qquad [1]$$

where a is the number of species present in both Q1 and Q2, b is the number of species present in Q1, and c is the number of species present in Q2. Values of the index can range from 0 to 1 and are expressed as percentages between 0 % and 100 %.

Unusually large changes in ground surface height were recorded, between 1996 and 2006, for three of the 1,836 survey positions These points stood out as clear outliers when the frequency distribution of change values was plotted. At least two of the three survey positions were in or on the sides of drains, where a slight horizontal offset can cause a large vertical change. These points had an undue influence on mean values of change so were excluded when the mean and its confidence interval were calculated.

RESULTS

Water table

Mean daily rainfall for April to September was 2.3, 2.9, 2.3 and 2.1 mm for the first, second, fifth and tenth years post-restoration, respectively. During the first year post-restoration, both the level (Figure 3a) and the range (Figure 3b) of fluctuation of the water table were intermediate between those of the control and those of open-bog reference areas. The treatments involving a combination of felling the

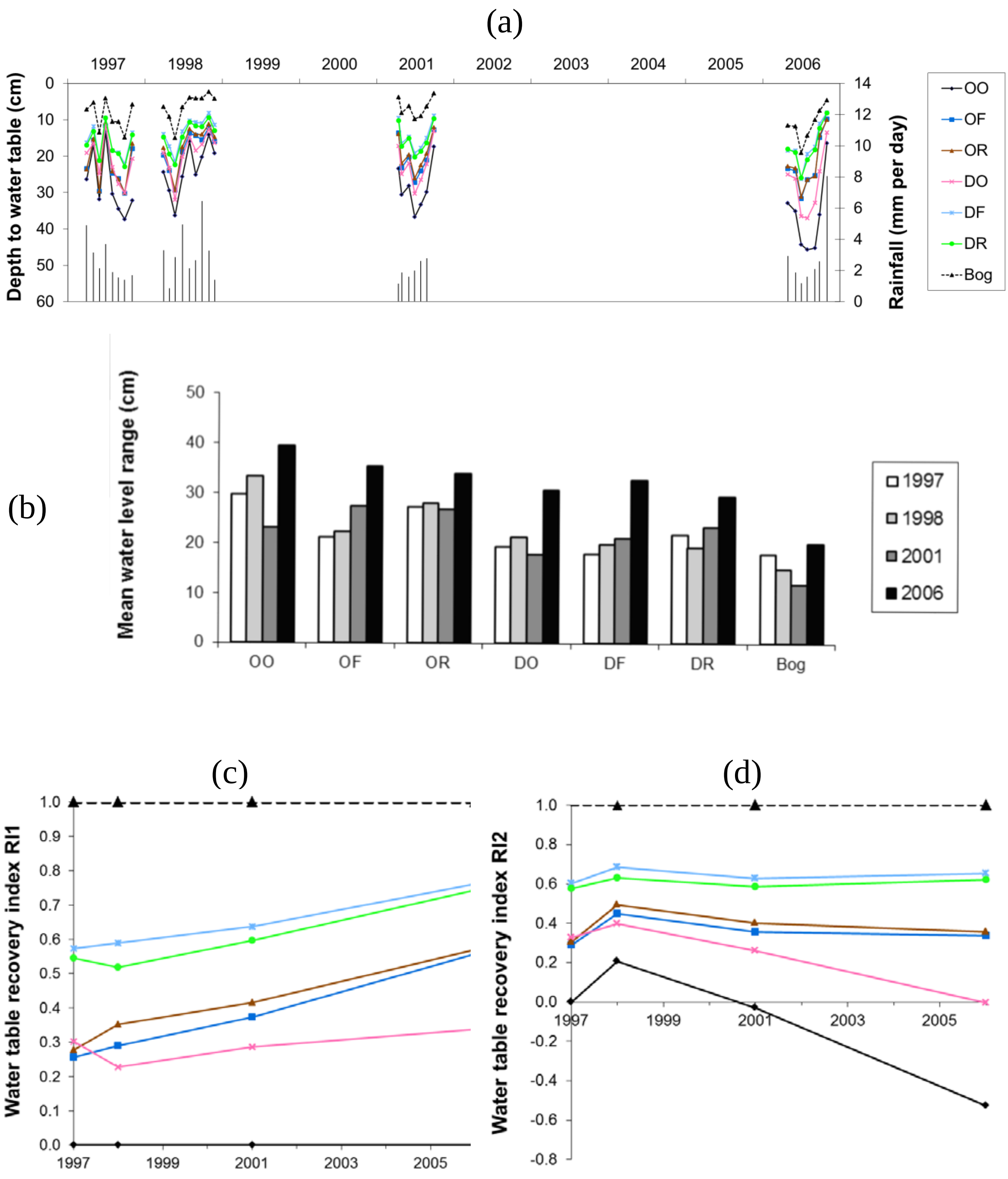

Figure 3. (a) Depth to water table for each treatment during the first, second, fifth and tenth growing seasons after applying the restoration treatments. Upper line: open bog adjoining the plots. Lowest line: control treatment, in which the trees continued to grow. The vertical bars represent the mean daily rainfall between dipwell readings. (b) Annual range of fluctuation of the water table by treatment and year using water level range gauges (walrags). The range for the undisturbed bog is shown for comparison. (c) Water table recovery index, RI1, which scales the water level for each treatment relative to that for the control (OO) (RI1=0) and the undisturbed bog (RI1=1). (d) Alternative recovery index, RI2, which scales the treatment water table using the 1997 (Year 1) control level as the baseline (RI2 = 0) and each year's reference bog level as the ceiling (RI2 = 1). The key to treatments for (c) and (d) is the same as for (a).

trees and damming the furrows (DF and DR) came closest to the reference level. By the tenth year post-treatment, water levels in the treatments involving felling (OF, OR, DF and DR) remained 5–10 cm below the open bog reference, while water levels in the control were deeper than they had been previously. The annual range of fluctuation, based on the walrag minima and maxima, was smallest for the treatments involving damming plough furrows (DO, DF and DR) but not as small as that of open bog (Figure 3b). Changes in fluctuation range over time in the open bog, particularly the increased range in 2006, are thought to be partly due to variability in weather conditions from year to year.

The water table recovery index RI1 increased during the ten years for all the restoration treatments (Figure 3c), but the change was generally not significant ($p = 0.129$) and was smallest for DO. However, this improvement was relative to the control, in which the water table fell over the ten years and showed continued evidence of drying and subsidence as the trees continued to grow. Using RI2 to avoid comparing the treatments to a control that was in fact changing over time, RI2 increased in Year 2 ($p = 0.021$) in all treatments because of the very wet growing season. Continued tree growth in the control caused further lowering of the water table so that by Year 10 it had an RI2 of around -0.5, indicating that it had dried out further ($p = 0.21$). In Treatment DO, the water table initially showed partial recovery in response to furrow damming but then this recovery reversed and the treatment ultimately failed to raise the water table sufficiently to kill the trees within ten years. Treatments OF and OR recovered very well initially (RI = 0.6) but failed to recover further in Years 5–10. Only Treatments DF and DR made a further recovery during the period 5–10 years from the application of the treatments.

At Year 10, an internal edge effect was evident inside the forest margin in the control (OO) (Figure 4). Besides a step change in depth to water table at the forest edge, which was particularly pronounced in the driest conditions, depth to water table increased with distance from the forest edge under the trees for at least 45 m. This internal edge effect was modified by the restoration treatments. In treatment DO, where the forest edge remained at its former position, the step change at the edge had diminished, the level deepened with distance inside the forest margin and there was only a small step change at the inner margin of the treatment plot. In the other treatments, the former internal edge effect had become external to the new forest edge. The step change at the former forest edge had disappeared and there was a gradual increase in depth to water table with distance in from the former forest edge across the restored area to the new forest edge, where a reduced step change was evident.

Vegetation

The most striking change in the felled treatments was the rapid recovery of *Eriophorum vaginatum* (hare's-tail cottongrass) (Figure 5). The species composition of the vegetation in the control (OO) and

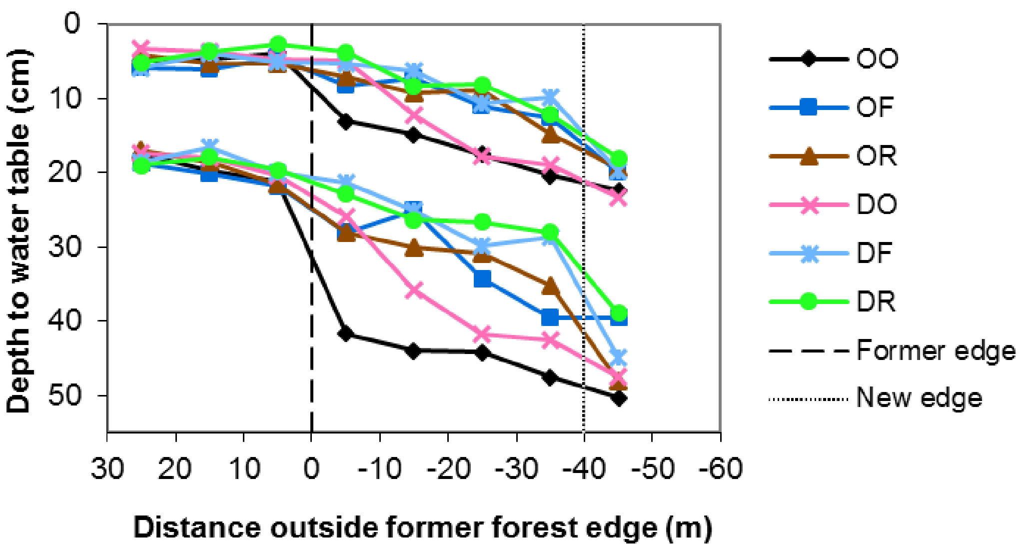

Figure 4. Mean depth to water table for each treatment ten years after restoration on a transect from open bog (left), across the treatment plot and into the adjacent forest (right). The upper set of lines is for the reading following the wettest period (02 Nov 2006) and the lower set is for the reading following the driest period (04 Jly 2006).

(a)

(b)

Figure 5. Treatment R: visual change in vegetation between (a) Year 1 and (b) Year 4.

the treatment that did not involve felling (DO) remained unchanged and were similar, as would be expected. Their species composition also remained distinct from that of the unplanted bog reference areas ten years after the treatments had been applied, as illustrated by the loci of quadrats on a correspondence analysis ordination (Figure 6). A difference between sites is, however, evident for the non-felled treatments (OO and DO) and is presumably due to the difference in tree species and age (Table 1).

In the felled treatments (OF, DF, OR and DR) the species composition had reverted part-way towards that of the unplanted bog reference areas (Figure 6). The loci for OR and DR were more or less identical, as were those for OF and DF, provided the two extreme outliers were omitted.

Over the ten years from the application of the treatments, plough-ridge plant communities of the four felling treatments (OF, OR, DF and DR) (Figure 7a) progressed towards the unplanted bog reference with some convergence, while ridges in the control and dam-only treatments (OO and DO) diverged from those of the felling treatments and barely progressed towards the reference. Vegetation on the original surface in the four felling treatments also progressed towards the reference but treatment DF diverged from the others (Figure 7b). In the dam-only and control treatments (DO and OO), original surface vegetation deviated from the reference, diverging from that of the felled treatments. Plough-furrow vegetation deviated from the reference in all the treatments but progressed very slightly towards the reference in the control (Figure 7c).

Species composition in the control and all the treatments (Figure 8) initially became increasingly different from the unplanted bog reference vegetation. This trend continued in the control (OO) and dam-only (DO) treatments whereas in the felled treatments there was a tendency towards increased similarity with the unplanted reference bog vegetation from Year 2. At Year 10, similarity to unplanted bog was higher ($p = 0.008$) for the felled treatments (mean Sorensen Index 41 %) than for the dam-only treatment (29 %) and control (24 %). These similarities were much lower ($p = 0.0000009$) than those among quadrats on unplanted bog adjoining the plots, where local variation resulted in a mean

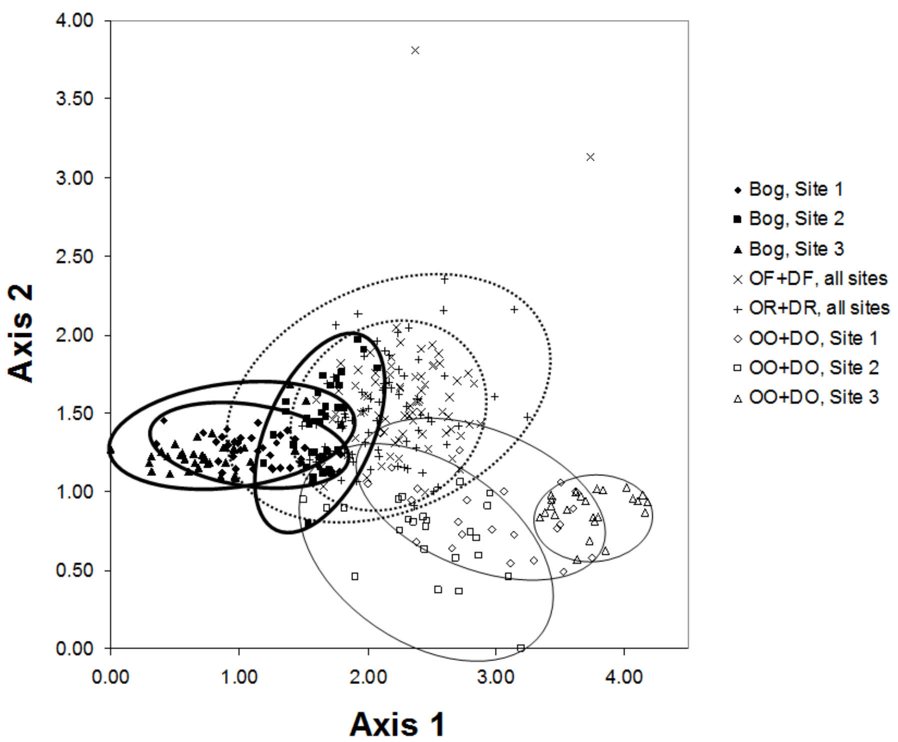

Figure 6. Vegetation groupings in a correspondence analysis ordination of individual quadrats based on cover of the species present ten years after restoration. Each ellipse includes all the quadrats of the grouping it represents, except the ellipse representing the OF and DF treatments, from which two extreme outliers have been omitted. Ellipse boundaries: bold = existing bog sites, dotted = felled/removed sites, weak solid line = control & blocked sites with trees left growing.

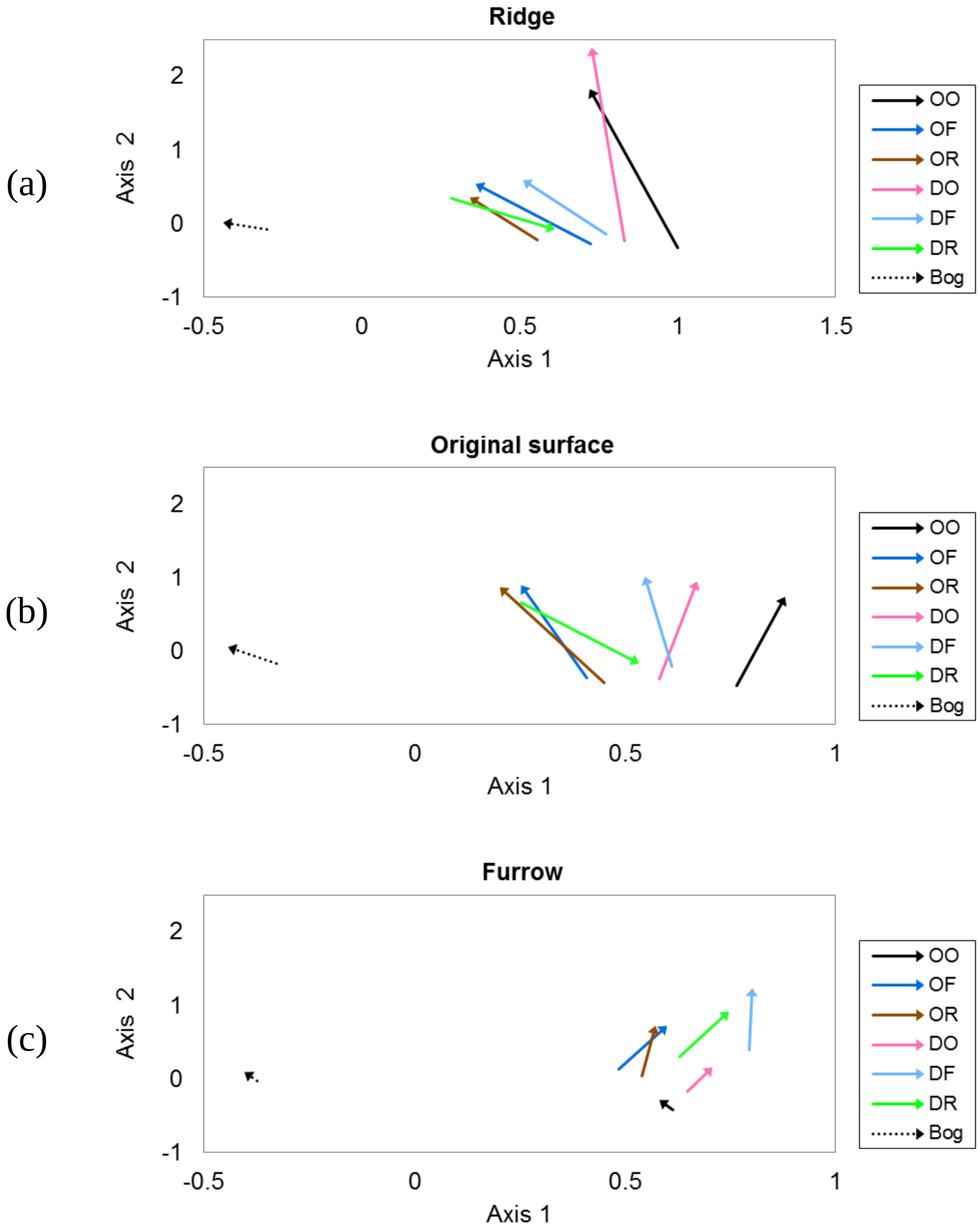

Figure 7. (a) Vegetation composition trajectories for the restoration treatments, the control and the undisturbed bog reference areas. (a) Plough ridges, (b) the original surface and (c) plough furrows. These represent the change between Year 1 and Year 10, with the arrow showing the direction of change.

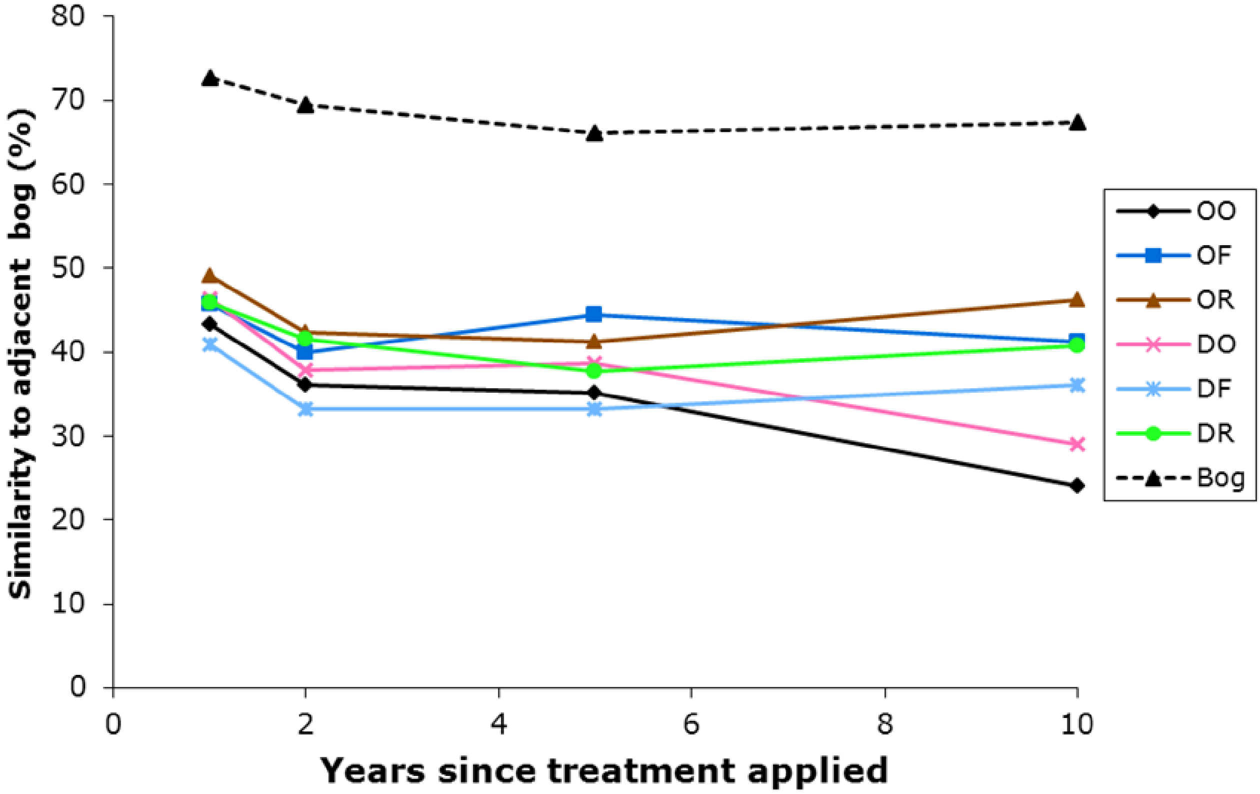

Figure 8. Vegetation recovery over time following restoration. The Sorensen Index was used as a measure of how similar the species composition of vegetation in the various treatments was to that of the adjoining undisturbed bog. Similarity among quadrats on undisturbed bog is included to show the inherent variability.

Sorensen Index of 67 % among quadrats on bog adjoining the same plot (*i.e.* 10 m or 20 m apart).

Vegetation differentiation at Year 10 between the plough furrows and both the original surfaces and the plough ridges was lower ($p = 0.03$) in the control, dam-only and felled un-dammed treatments (OO, DO, OF and OR) (mean Sorensen Index = 58 %) than in the felled and dammed treatments (DF and DR) (mean Sorensen Index = 40 %).

Tree regeneration from seed occurred in all treatments and was predominantly of lodgepole pine, with a very low density of Sitka spruce (Figure 9a). There were no significant treatment differences. Mean seedling density fell from 1100 *per* ha in the first year after treatment to 400 *per* ha the next year and then built up to a constant density of around 2700 *per* ha after nine years. Seedlings continued to grow in height and, after ten years, were equal - in terms of summed height *per* hectare - to a three-year-old planted conifer crop (Figure 9b). A regeneration edge effect was seen in all treatments, with seedling density greatest near the inner margin of each treatment plot (where it adjoins forest) and reducing with distance out from this margin (Figure 9c).

Ground surface height and peat properties

The ground surface in the control (OO) continued to subside as the trees continued to grow (Figure 10a). In contrast, the restoration treatments that involved damming the plough furrows (DO, DF and DR), caused the ground surface to rise by 5–7 cm over ten years and this was significant at $p < 0.05$. On the adjacent unplanted bog outside the original forest edge, the ground rose in all treatments including, surprisingly, the control (Figure 10b).

The peat water content of both the 10–20 cm and the 70–80 cm depth layer increased ($p = 0.000002$ and $p = 0.067$, respectively) over the ten years after the treatments were applied, in all treatments including the control (Figure 11a). However, during the second half of this period (from Year 6 to Year 10), peat water content decreased in the control treatment (OO) whilst in most of the restoration treatments it continued to rise.

Peat dry bulk density decreased in both 10–20 cm ($p = 0.0011$) and 70–80 cm ($p = 0.126$) depth layers over the ten years following treatment (Figure 11b). The decrease was greater in the 10–20 cm layer than in the 70–80 cm layer ($p = 0.030$).

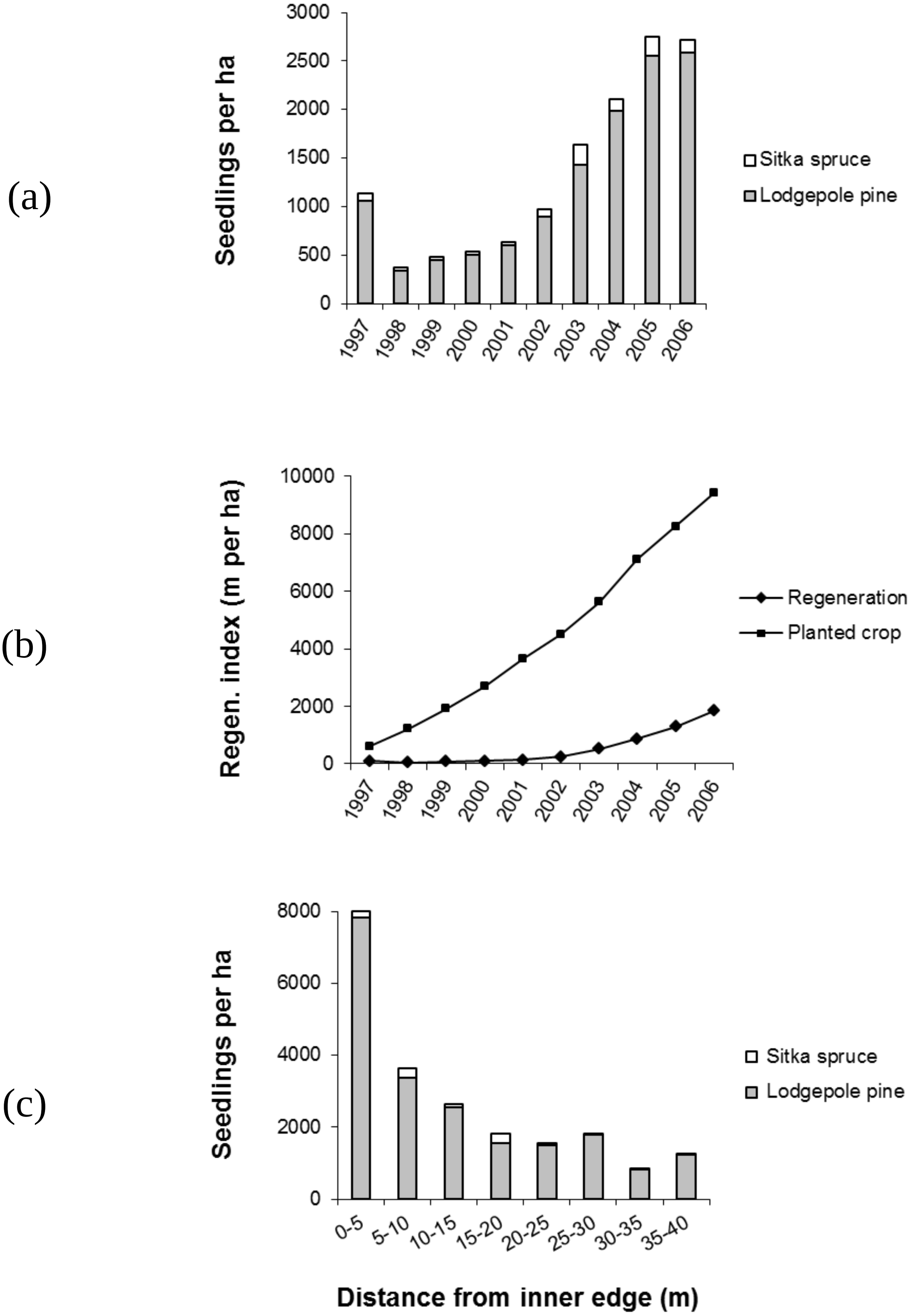

Figure 9. Tree regeneration in the felled treatments: (a) mean density over the four felled treatments (OF, OR, DF and DR) ten years after restoration; (b) regeneration index (summed height *per* ha) for each treatment over ten years compared to that of a planted conifer crop on a similar site nearby; (c) distribution of regeneration across the treatment plots, from the inner edge adjoining forest (at 0 m) to the outer edge adjoining unplanted bog (at 40 m).

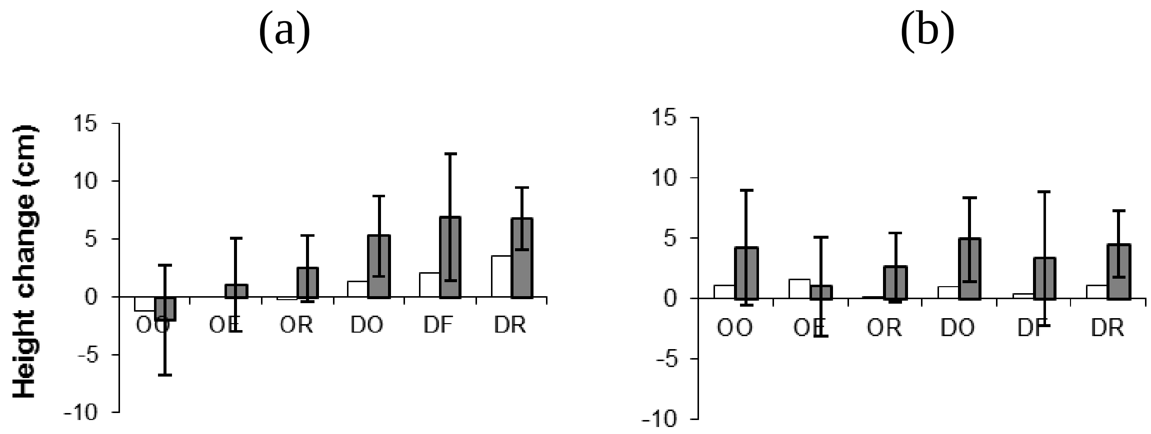

Figure 10. Five-year and ten-year change in ground surface height (unfilled and filled bars, respectively) in response to the different treatments for (a) the section of the monitoring transect within the treatment plot, and (b) the adjacent open bog 1–10 m outside the former forest edge. The error bars are 95 % confidence intervals (n = 6).

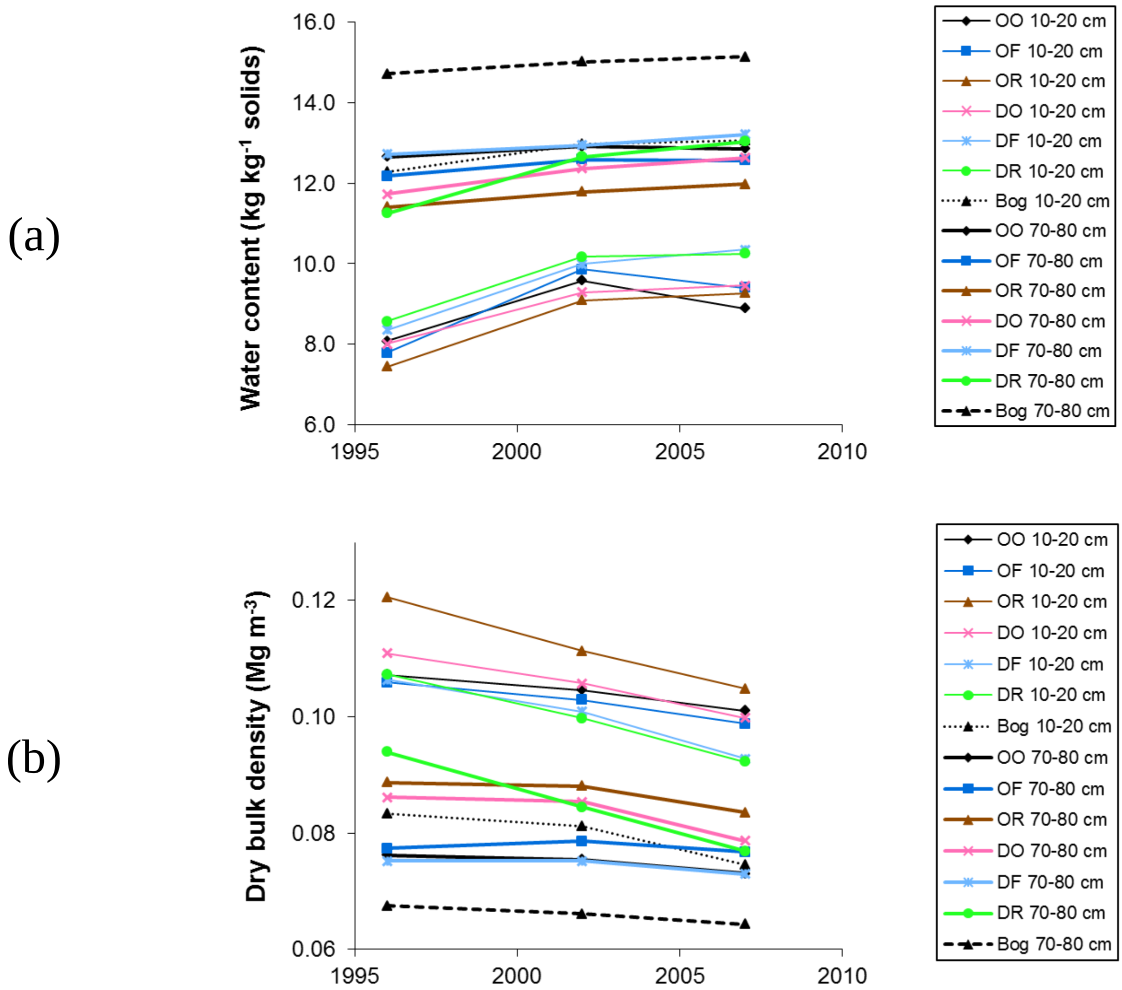

Figure 11. Ten-year change in (a) water content and (b) dry bulk density of peat at two different depths in response to the restoration treatments. Mean values for samples from the adjacent bog are shown for reference.

DISCUSSION

Water table response to restoration

The results illustrate the difficulty in detecting genuine changes in water level and how easily incorrect conclusions can be drawn. A control was needed to demonstrate the effects of restoration treatments. However, because monitoring had to continue for many years to detect effects related to gradual vegetation change, the water level in the control fell as the trees continued to close canopy and grow. By comparison with the control, it appeared that the water table had made a secondary recovery in the restoration treatments (Figure 3c) but in fact this was due mainly to a further fall in water level in the control (Figure 3d). After an initial rapid partial recovery of the water table in all treatments, only in treatments DF and DR was there continued recovery between Years 5 and 10, and this was only slight. There were varying degrees of deterioration in the other treatments. This may be an important indication of longer term outcomes. It is not known to what extent the recovery of the water table was limited by proximity to the new forest edge but it is important to recognise this potential limitation.

Water table behaviour models may offer a sensitive means of detecting changes in water table behaviour resulting from restoration activities. Rennolls *et al*. (1980) proposed a very simple model that could be further developed for this purpose. Fitting the model to a time series of water-table levels before and after restoration would allow values of inherent non-weather-dependent soil water parameters to be estimated. The changes in the parameter values could aid understanding of water table regime change and allow prediction of its fluctuations under future precipitation scenarios.

Forest evapotranspiration plays an important role in lowering the water table for forestry on peatland (Sarkkola *et al.* 2010, Sarkkola *et al.* 2013) and rises in water table level in response to forestry clear-felling have been widely reported (*e.g.* Pyatt *et al.* 1985, Roy *et al.* 1997). Haapalehto *et al*. (2011) report a rapid initial rise of the water table, similar to that reported here, after felling trees and blocking drains at forestry-drained low-sedge bog and pine fen sites in southern Finland. There was some further recovery of the water table during the second and third years after restoration but by Year 10 it had fallen slightly, although that might have been due in part to the low rainfall that year. Unlike the Scottish sites where the water table continued to fall in the control, at these Finnish sites the water table rose over ten years in the drained controls, and this was speculatively explained by the authors as being due to natural infilling of the ditches by *Sphagnum* growth. The difference probably reflects a less dense forest canopy at the Finnish sites than at the Scottish sites, where complete canopy closure reduced light levels, limiting *Sphagnum* growth.

A question that arises is whether the partial water table recovery reported here is sufficient to restore the valued ecosystem functions of blanket bog, particularly its provision of habitat for specialist plants and animals and its net climate cooling effect *via* carbon sequestration and avoided green-house gas losses. Our reference sites were on open bog adjacent to the former forest edge and had not been ploughed or planted. They may have been subject to a forest edge effect but we think that any such effect would have been small. For example, Shotbolt *et al*. (1998) reported subsidence extending on average 30 m from the edge of 28-year-old forest plots at Bad a' Cheo (also in Caithness), whereas the forests in our study were only 11 and 16 years old when felled. The depth to water table for the reference sites ranged between 3 and 19 cm. The equivalent range for the best restoration treatment (DF) was 8–26 cm, but for much of the growing season it was below 19 cm and thus below the lowest level in the reference sites. We can speculate on the implications of incomplete water table recovery for the condition of the vegetation. Some of the plants that occur in blanket bog vegetation, such as *Calluna vulgaris*, *Deschampsia flexuosa* and *Sphagnum capillifolium*, which are also found in drier habitat types such as acid grassland and dry heath, are likely to survive and perhaps thrive in restored bog. More hydrophilic species such as *Sphagnum papillosum*, *S magellanicum* and *S. austinii* may not survive or may persist only in the wettest microsites. It will be interesting to see whether the water table recovers farther in the longer term and, if not, how the vegetation composition develops in response to this incomplete recovery of the water table.

For the ten-year post-restoration period the hydrological edge effect, *i.e.* slight drawdown of the water table 5 m outside the former forest edge during dry weather, continued despite the restoration treatments (Figure 4). However, the step change in water table level between 5 m outside and 5 m inside the former forest margin greatly decreased with restoration. The step ranged from 1 cm to 6 cm in the restoration treatments compared to 21 cm in the control. Shotbolt *et al*. (1998) reported that the water table on open bog adjoining forest plots sloped down towards the plots but that, due to subsidence, the ground surface also sloped down with the result that depth to water table did not increase significantly with proximity to the forest. Reduction of the step change in depth to water table at the former forest edge by the restoration treatments in this experiment may have reduced further subsidence outside the former forest edge.

Vegetation response to restoration

Within 3–4 years, the vegetation of the restored plots superficially resembled the reference vegetation, mainly due to the rapid recovery of *Eriophorum vaginatum*, which is often dominant in undisturbed blanket bog. However, in terms of its species composition, it only partly reverted to that of unplanted bog in ten years (Figures 6 and 7). Studies of vegetation succession following clearcut on forestry-drained peatland in Finland have reported a rapid increase in cover of *E. vaginatum* on bare peat microsites produced by an excavator in preparation for forest regeneration, particularly scalps (patches of ground with the vegetation scraped off) but also peat mounds (Saarinen *et al*. 2009).

Plant community trajectories are sometimes used to show directions of change following restoration action (*e.g.* Haapalehto *et al*. 2011) but because they show multiple points in a time series, they become cluttered if multiple sites or treatments are plotted together. Matthews & Spyreas (2010) proposed a framework for using such trajectories to monitor wetland restoration. They suggested interpreting the successional direction after restoration in terms of community convergence or divergence and progression towards or deviation from reference sites. This helps, but progression towards the reference in one dimension, for example that representing Axis 1, can be accompanied by deviation away from the reference in another dimension, for example Axis 2, giving a confused picture. Our interpretation of trajectories representing the ten-year change resulting from our treatments (Figures 7a–c) is that ridges and the original surface are responding to the restoration treatments rather slowly but in the right successional direction, whereas plough furrows are succeeding towards a different plant community from that found on unplanted bog. This may be because the furrows are deeper and steeper-sided than the natural small depressions found on undisturbed bog, with higher shade levels and nutrient concentrations playing a part.

The vegetation in plots that were felled became more similar to the vegetation of unplanted bog from Year 2 (Figure 8). In treatments OR, DF and DR, the increased similarity was consistent up to Year 10, suggesting that the water table had recovered sufficiently to drive the recovery of bog vegetation. Treatment OF is more doubtful because a strong increase in similarity to unplanted bog between Years 2 and 5 was followed by a decrease between Years 5 and 10.

Natural regeneration of trees on restored afforested peatland can occur by regrowth of cut trees (*i.e.* from branches below the cut) or from seed. In our experiment, conifer regeneration from seed affected the treatments in which the trees had been felled (OF, OR, DF and DR), reaching a density that would eventually lead to forest regrowth if left uncontrolled. However, the density decreased sharply with distance out from the new forest edge, a finding consistent with the distribution of conifer regeneration reported for undisturbed blanket bog (Manzano 2012). Because the entire area of our small plots was within 40 m of a remaining forest edge, they were strongly affected by windblown seed. A lesser amount of regeneration from seed would be expected in larger-scale restorations but a zone of dense regeneration might always be expected close to any remaining forest. Further research is required to determine whether timing and methods of forest clearance affect subsequent tree regeneration and, if so, to develop methods that minimise the need for control operations. It is possible that where forests are cleared by machines (*e.g.* felling by harvester and/or extraction by forwarder) the resulting ground disturbance might increase tree seedling establishment by increasing the occurrence of favourable microsites, such as poorly decomposed bare peat (Groot & Adams 1994) or patches of *Sphagnum* (Saarinen 2002).

Reversal of peat subsidence by restoration

It is well known that afforestation of deep peat can cause subsidence of the ground surface (Pyatt *et al*. 1992, Shotbolt *et al*. 1998, Anderson *et al*. 2000) although the extent to which this is due to peat shrinkage, consolidation under compression and oxidative wastage has not yet been established. The limited literature on peat shrinkage suggests that shrinkage due to drying is irreversible but a limited amount of rebound can occur when compression is relieved (Hobbs 1986). Our results demonstrate some reversal of subsidence if the furrows were dammed. This is most likely to be due to rebound from the compression caused by drained peat weighing down on the saturated peat below the water table. The significant furrow-damming effect supports this but the fact that the surface rise occurred gradually over ten years, rather than rapidly in the first year, shows that rebound of compressed peat is a slow process. This is not surprising, given that secondary compression of peat is itself a slow process. Our surveying method was intended to measure changes in the soil surface level rather than any vegetation surface, but we cannot rule out the possibility that thickening of the bottom layer of vegetation contributed to the surface rise. However, vegetation thickening would not explain the significant furrow-damming effect.

A decrease in dry bulk density and increase in water content of the peat occurred in the restoration treatments, which is consistent with rises in water table level and ground surface level in these treatments. However, in the control (OO), where the ground surface level fell, the peat bulk density also decreased over the ten years, while water content increased over five years before decreasing over the next five years. This was a surprising result and one that we cannot explain. One possibility is that the water table in the control rose during the time that elapsed between the treatments being applied (Oct–Dec 1996) and the start of water table monitoring (Apr 1997), perhaps due to an edge effect from the adjacent restoration treatments.

The restoration treatments also caused some rebound of the peat on bog adjoining the former forest edge, suggesting that subsidence of both the afforested ground and the adjacent non-afforested bog are partly reversible. Reduction of any slope towards the forest resulting from subsidence is important in limiting the extent of the hydrological sink effect around forests on flat blanket bog. The rise in the surface level of unplanted bog adjoining our controls led us to speculate that compression under the forest may sometimes cause uplift on adjacent ground that is not subject to the overburden pressure resulting from the weight of a drained peat layer and a growing stand of trees, a possibility also suggested by Shotbolt *et al.* (1998).

Success of the restoration treatments

The success of our restoration treatments reflects the combined responses of the water table, vegetation, ground surface and peat properties to the treatments. None of the treatments can be said to have already restored the blanket bog habitat because they have not yet fully reversed the changes in water-table level and vegetation composition caused by afforestation. Treatments DF and DR, which involved felling the trees and damming the plough furrows, look likely to succeed but that depends on whether the water table continues the very slow secondary recovery it made between Years 5 and 10. Treatments OF and OR may also be on course to succeed but their vegetation recovery is more evident than their water table recovery, which seems not to have continued after the first five years. If the vegetation continues to become increasingly similar to the bog reference vegetation and the water table remains within its current range, full restoration to blanket bog may occur. Where lowered water tables persist, however, these restoration areas may be vulnerable to drought in dry summers and remain degraded. Treatment DO resulted in no significant change for the forested bog because damming the furrows failed to kill the trees. Consequently, light was not re-admitted to the ground surface and the water table remained lowered.

CONCLUSIONS

- To restore afforested blanket bog it is necessary to fell planted conifers. Ditch blocking alone is unlikely to kill them.

- It is not necessary to remove pre-commercially felled trees unless there are other reasons for doing so. Bog vegetation can develop even if they are left on the ground.

- Damming drains and plough furrows helps to raise the water table and may help to provide aquatic microhabitats.

- A combination of felling trees and damming drains and plough furrows looks likely to restore the former wildlife habitat and carbon sink functions of blanket bog.

- Natural regeneration of trees can occur. If this regeneration is from seed, control measures are likely to be needed where self-seeding is densest, near remaining areas of forest. Further research is needed to determine whether timing and method of felling can be optimised to reduce conifer regeneration on restored afforested bogs.

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