

An experimental study on the response of blanket bog vegetation and water tables to ditch blocking

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Abstract We studied the effect of ditch blocking on vegetation composition and water-table depths in a blanket peatland. Measurements were made for a period of four years (water tables) and five years (vegetation) in the inter-ditch areas of three experimental treatments: (i) open ditches, (ii) ditches blocked with closely-spaced dams and (iii) ditches partially infilled with peat and blocked with dams. It is often assumed that ditch blocking will lead to an increase in the abundance of *Sphagnum* and, potentially, a reduction in the abundance of sedges, particularly the cotton grasses. However, our data show no treatment effects on the abundance of either group. We did find an effect of time, with the abundance of both sedges and *Sphagnum* spp. varying significantly between some years. For the sedges there was no systematic change over time, while for the *Sphagnum* spp. abundance tended to increase through

the study period. This systematic change was not related to a measure of the vigour of the sedges, although vigour was lower towards the end of the study compared to the beginning. Our vegetation data are consistent with our water-table data. As with plant type abundance, we did not find any statistically significant differences in water-table depths between treatments, both for annual averages and summer averages. We comment on why ditch blocking does not seem to have affected water tables and vegetation composition at our study site.

Keywords Blanket peatland · Re-wetting · Vegetation response · Cotton grass (*Eriophorum vaginatum*) · *Sphagnum* · Hydrological response

Introduction

In this paper we consider how restoration of drained blanket peatlands affects the composition of their vegetation. Although globally rare (Gallego-Sala and Prentice 2012), blanket peatlands are the most important peatland type in the UK, both by area (15,736 km²—c. 87% of the total UK peatland area—Baird et al. (2009)) and carbon (C) stored (conservatively, at least 1000 Mt C—see end note). Large areas of these peatlands have been ditch drained, with drainage carried out to improve the grazing potential of the land (Holden et al. 2004; Parry et al. 2014). Because of

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concerns that drainage has damaged the conservation value and C storage potential of these peatlands, there have been calls to undertake large-scale damming or infilling of drainage ditches as a restoration measure (Bain et al. 2011). Despite such calls, there is a general lack of information on how peatland water tables respond to ditch blocking and how blocking affects the composition of blanket peatland vegetation (Parry et al. 2014).

Stewart and Lance (1991) investigated the effect of open ditches, but not blocked ditches, on both water tables and vegetation on blanket bog at Moor House in the N Pennines, N England. The ditches they investigated were roughly contour-parallel. They took a long-term perspective (>20 years) on how drainage affects vegetation and also undertook a short-term (<2 month) study on water-table behaviour between the ditches. They found that, within 20 years of drains being cut, *Sphagnum capillifolium* (Ehrh.) Hedw. had disappeared from ditch edges and virtually disappeared from inter-ditch areas away from the edges. For other blanket peatland plant species the effects of ditches appeared to be more local to the drain or to change over time since drainage. For example, in an apparent response to drainage, the cover of *Calluna vulgaris* (L.) Hull increased and peaked at eight years but later declined to become statistically insignificant. In their hydrological study (summer 1979), Stewart and Lance (1991) found that average water-table depths varied spatially by as much as 15 cm, with the deepest water tables close to the ditches and shallowest towards the mid-points between the ditches. The hydrological data collected by Stewart and Lance (1991) confirmed that at least 20% of the area between drains was affected by the drain (i.e., was drier than would be the case in the absence of the drain). Their *S. capillifolium* data accord with the hydrological data in that the species disappeared from drain edges where drainage effects were greatest; however, that it also declined substantially in the inter-ditch areas away from drain edges may suggest that drains caused all of the inter-drain area to be drier than would be the case naturally.

In a study of drainage and sheep grazing effects on the abundance of different plant species within a Welsh blanket peatland, Wilson et al. (2010) found some evidence that blanket bog species such as *Eriophorum angustifolium* Honck were less abundant in a zone extending up to 15 m from drainage ditches

but found that drainage effects were partly confounded by the effects of sheep grazing. Wilson et al. (2011) note that their vegetation results are consistent with other work at their study site (Wilson et al. (2011)—see below) that shows that ditches lower water tables in their vicinity.

On a blanket peatland in NE Scotland, Bellamy et al. (2012) looked at drainage and re-wetting effects on vegetation composition using two sites where ditches had been blocked and two where the ditches remained open. Unlike the ditches investigated by Stewart and Lance (1991), the ditches studied by Bellamy et al. (2012) mostly ran downslope. Vegetation composition and cover were measured in transects perpendicular to the long-axes of the ditches. However, while one blocked site showed the patterns expected—more wet indicator species and fewer dry indicators than the open-ditch sites—the other blocked site did not. Bellamy et al. (2012) did not look at water tables at their sites, so it is unclear if the apparent inconsistency between the two blocked sites was caused by hydrological differences.

Most recently, Williamson et al. (2017) investigated vegetation changes in response to drain blocking in a blanket peatland in N Wales (in an area adjacent to the one used for the current study—see “[Materials and methods](#)” section below). They found no effects of blocking on vegetation composition and suggested that peat subsidence may explain their findings (see “[Discussion and conclusion](#)” section below). They undertook a one-off vegetation survey of the peatland next to open ditches and next to ditches that had been blocked for one, four and five years. Because they did not measure vegetation changes over time they were unable to determine whether factors related to the year in which they did their vegetation survey (e.g., an unusually wet year prior to the survey) masked any effects of drainage.

In all of the above studies, the inter- or between-ditch areas were investigated. In a study of greenhouse gas fluxes from a Welsh blanket peatland, Cooper et al. (2014) noted a distinct change in vegetation over a 27 month period within blocked (infilled) ditches. Over this time *Eriophorum vaginatum* L. colonised and came to dominate the within-ditch area. However, changes in *E. vaginatum* abundance were not monitored quantitatively; i.e., numerical estimates of abundance were not made. Peacock et al. (2013) measured changes in the vegetation cover of pools

formed by dams on another blanket peatland site in N Wales (the one used for the current study—see “Materials and methods” section below). Eighteen months after their formation, pools had been colonised by a mixture of *E. vaginatum* (average of 37% cover) and *Sphagnum cuspidatum* Ehrh. ex Hoffm. (38% cover) ($n = 60$ pools). Like Cooper et al. (2014), Peacock et al. (2013) did not look at the vegetation in inter-ditch areas.

Work has also been done on drain-blocking effects on the vegetation of both raised bogs and fens, with examples including Komulainen et al. (1999) and Haapalehto et al. (2011). However, it is difficult to gauge how relevant the findings from these studies are to blanket peatlands. Key peat properties may differ between peatland types, and raised bogs, for example, appear to have a higher permeability (hydraulic conductivity) than blanket bogs which will affect the degree to which drains and drain blocking influence water tables (Evans et al. 2014) and, therefore, vegetation.

A number of studies have been conducted on how drain blocking affects blanket peatland water tables. Examples include Holden et al. (2011) and Wilson et al. (2010). Both considered drains that run cross slope and both found the effect of ditch blocking on water tables was statistically significant but relatively modest—of the order of 2–3 cm. Whether such changes are sufficient to cause changes in vegetation composition are unclear. As noted above, for the same sites reported in Wilson et al. (2010, 2011) found some effect of drains on the composition of blanket peatland vegetation, although this finding was partly confounded by sheep grazing effects. This finding suggests that small changes in water-table depth can cause changes in vegetation composition. However, studies on the hydrological regimes of raised and blanket bogs by, respectively, Belyea (1999) and Laine et al. (2007) suggest that water-table depths in different microhabitats or vegetation types vary by at least 5–10 cm. For example, Laine et al. (2007) found that average water-table depths consistently differed by 5–10 cm between lawns and hollows (for microhabitat nomenclature see Belyea and Clymo (2001)) and by 10–15 cm between lawns and hummocks. Therefore, changes in mean water-table depths of a few cm from drain blocking may be insufficient to cause substantial changes in vegetation type or of the relative abundance of a particular vegetation type’s

component species. Despite such uncertainties on drain blocking effects, some prominent reviews such as Lindsay (2010) suggest that ditch blocking on blanket bogs will lead to an increase in the abundance of *Sphagnum* mosses at the expense of cotton grasses. However, there is a lack of data on ditch blocking effects; in particular, there is a lack of fully replicated experimental studies where both vegetation and water tables are studied concurrently and changes over time measured. To help address this research gap we conducted field trials on a UK blanket peatland. We studied the response of *Sphagnum* spp. and *Eriophorum* spp. over a five-year period after ditch blocking, and monitored water tables for the first four years. We focused on the inter- or between- ditch zone because, by area, this is the largest part of any ditch-blocked peatland.

Materials and methods

Study site and meteorological conditions

The field trial was carried out on part of the Migneint blanket bog in N Wales, UK (52.97°N, 3.84°W). The field site was located near the top of a north-facing hillslope at c. 500 m above sea level, with a total area of c. 2 ha drained by a set of 12 parallel ditches running approximately downslope, which are believed to have been dug in the 1980s (Fig. 1). The ditches had a mean spacing of 16 m (range 11–26 m), mean slope of 4.5° (range 3.9–5.1°), and a mean monitored length of 99 m (range 84–107 m). They were mostly shallow (50–70 cm deep) and in some cases overgrown by vegetation, although all were found to be hydrologically functional at the start of the experiment. Measured peat depth across the study area ranges from around 0.5 m to 2 m. The peat overlies Cambrian mudstones and siltstones (Lynas 1973) but these appear to be largely hydrologically isolated from the overlying peat. The pre-trial vegetation comprised a typical blanket bog assemblage including *Calluna vulgaris*, *E. vaginatum*, and various species of *Sphagnum*, although the abundance of the latter group was variable and sometimes low (see “Results” section).

In February 2011, eight of the 12 ditches were blocked using two different methods, with four open ditches retained as controls. Ditches were assigned to treatments by first placing them into three groups of

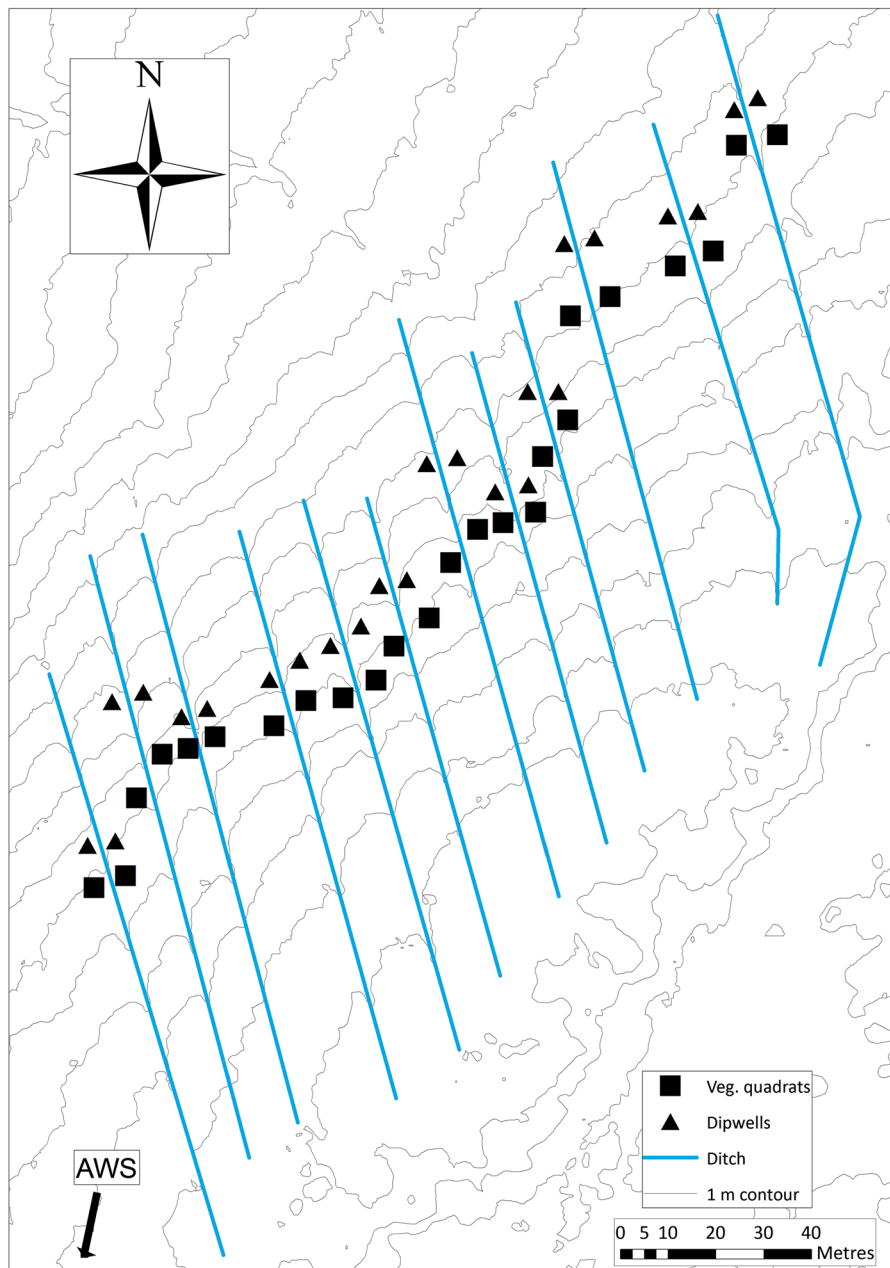


Fig. 1 Map of the experimental area. Meteorological data were collected using an on-site automatic weather station (AWS), the location of which is also indicated

four based on their measured pre-blocking discharge rate, and then randomly assigning one ditch from each subset into one of the three treatment/control categories. The two ditch-blocking methods used were damming, and reprofiling plus damming (henceforth referred to as ‘reprofiling’). For the dammed

treatment, peat dams were constructed at regular intervals along the ditch using peat extracted from ‘borrow pits’ adjacent to the ditch, creating a sequence of pools behind the dams. For ditches in the reprofiling treatment, ditch vegetation was removed, the peat base compressed to destroy any natural pipes (*sensu* Jones

1981) that may have been present, the ditch partially infilled with peat scraped from inter-ditch areas, and the vegetation replaced. This treatment also involved the construction of peat dams, but pools tended to be shallower or absent due to the partial infilling of the ditch.

Water tables and meteorological conditions were measured up to March 2015. The weather at the site (as measured using an on-site automatic weather station) was highly variable over this period and is summarised in Fig. 2. The annual mean air temperature was 7.8 °C (March 2011–March 2012–Year 1), 6.5 °C (2012/13–Year 2), 6.9 °C (2013/14–Year 3) and 7.6 °C (2014/15–Year 4). In the same period, the rainfall for each project year was 2255, 2409, 1786 and 1936 mm, respectively. Year 3 had a substantially drier summer (Jun, Jul, Aug) than the previous two years. The cold winter of 2012–2013 and very cold early spring of 2013 also stand out, as does the late winter and early spring drought and high temperatures of February–March 2012 (Fig. 2).

Vegetation sampling

The vegetation at the site was monitored using permanent 1 m × 1 m quadrats set up in 2010 before the ditch blocking took place (Fig. 1). Twenty-four permanent quadrats were established 2 m to the east and west of each ditch. These were placed away from those inter-ditch areas where peat was used for damming or infilling of the ditches (see above). Measurements took place once per year at the end of the growing season (late August to early October), and comprised the following:

- The abundance of all sedges (including the cotton grasses (*Eriophora*) and *Carex nigra* (L.) Reichard) and all *Sphagnum* spp. was estimated using a nested frequency measure. These species were chosen because they are believed to be important controls on methane (CH₄) emissions from peatlands (e.g., Lindsay (2010), Green and Baird (2012))—the wider study, of which the vegetation study was a part, was an investigation of how ditch

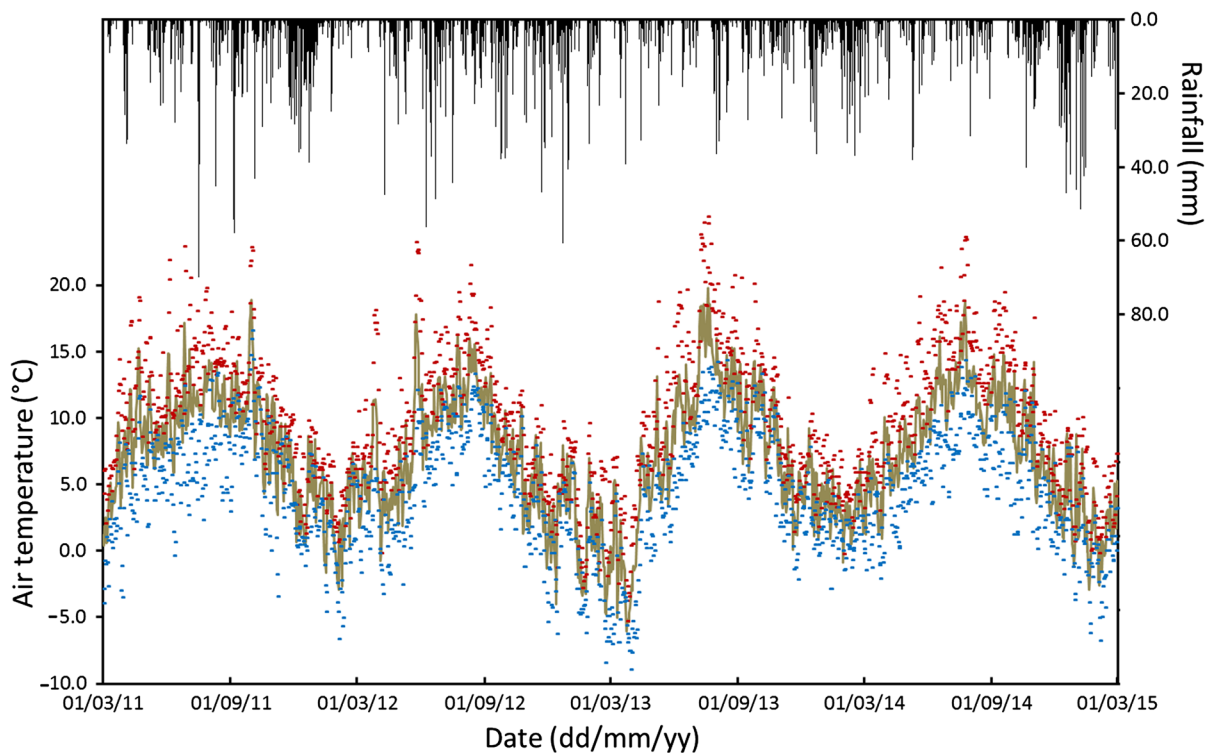


Fig. 2 Meteorological conditions at the experimental site between 1st March 2011 and 1st March 2015. The black bars indicate daily rainfall. Average daily air temperature is denoted

by the green–brown line, daily high temperature by the red dashes and daily low by the blue dashes

blocking affects blanket peatland CH₄ budgets. The measurement of nested frequency involved noting the presence or absence of each target sedge and *Sphagnum* species in each of the 100 10 cm × 10 cm squares comprising each quadrat. A species was marked as present even if it was not rooted in a 10 cm × 10 cm square. Using this method, a species present in, for example, 46 of the 100 squares, would have a nested frequency of 46% or 0.46. On a species by species basis, this method will give a value that is correlated to cover (and which becomes closer to cover as square size decreases). For any single species, or group of related species, it provides a simple and objective measure that allows abundance to be compared over space (between locations or management treatments) or over time (across successive years after management change). It is, however, unreliable for comparing abundance between different species. For example, a species like Cranberry (*Vaccinium oxycoccos* L.) has thread-like branches which often form a ‘net’ underneath taller shrubs. Depending on square size, a species such as this could produce a very high nested frequency even though its real cover was low. In contrast, tussocks of sedges such as *E. vaginatum* might have a relatively high true cover but occupy relatively few 10 cm × 10 cm squares. As such, it makes little sense to compare these species using nested frequency.

- The ‘vigour’ of the sedges within each quadrat was also recorded by measuring, for each sedge species, the height and maximum width of the ten leaves nearest the centre of the quadrat. Where there were fewer than ten leaves, measurements were still made (and the number of leaves noted).

We grouped the quadrat data on abundance into a sedge group and a *Sphagnum* group. The sedge group was, in practice, almost entirely made up of *E. vaginatum*, while the *Sphagnum* group comprised, in order of abundance: *S. capillifolium*, *Sphagnum fallax* (Klinggr.) Klinggr., and *Sphagnum papillosum* Lindb.. These three species have a tendency to occupy different microhabitats (Clymo 1973), so a case could be made for separating them in any analysis. However, we chose to group them into a single category for two reasons. First, all were relatively sparse at the

beginning of the project, and overall *Sphagnum* abundance can be taken as an indicator of site wetness (with wetter sites often having a higher overall *Sphagnum* abundance than drier sites). Secondly, although they can occupy different microhabitats in undisturbed sites, the conditions under which they grow well can be quite variable and will depend on facilitation by, and competition with, other species. Thus, while a species may be associated with a particular water-table regime in a natural site, it is by no means clear that the same species will reliably indicate that water-table regime when it colonises a restored site. Wheeler et al. (2002), for example, make note of a study that showed that *S. fallax*, which is usually associated with hollows and wet lawns, was capable of colonising ‘dried-out’ bogs, this being facilitated by a cover of *Polytrichum alpestre* Hoppe and *E. vaginatum*.

Water table measurements

Water tables were measured using dipwells every three to six weeks, with the higher frequency during the growing season. The dipwells were placed 2 m west and east of the ditches ($n = 24$ wells) and c. 5 m downslope of the permanent quadrats. They were made from 1 m lengths of polyvinyl chloride (PVC) pipe with an outside diameter of 32 mm and a 3.5 mm wall thickness. The pipes were perforated with 8 mm diameter holes drilled at 100 mm intervals along four lines running lengthwise along the pipe, with the holes in each line offset from those in neighbouring lines by 50 mm. The dipwells were tested for their response times and all showed a 90 percent recovery within 120 min during typical water levels, with most recoveries substantially quicker than this.

Data analysis

For the vegetation data we investigated whether blocking treatment (control or open ditches, dammed, or reprofiled), position (east or west of the ditch) and time (Years 0–5; 2010–2015) had any effect on the nested frequency of the sedges and the Sphagna. For each plant group, a factorial, repeated-measures ANOVA was used (in IBM SPSS Statistics version 23). The sedge data were transformed (to the power of four) because they failed the test for normality. Where

Mauchly's test indicated a violation of the sphericity assumption, changes were made to the degrees of freedom using a Greenhouse Geisser correction to reduce the chances of a Type I error. Where $p \leq 0.05$, Tukey's HSD (honest significant difference) test was used to identify which blocking treatments, sides (east or west) or times (year of survey) were significantly different from each other.

The sedge vigour data from the 24 quadrats were collated and graphed according to treatment, position, and year. As noted above, by far the most common sedge at the site was *E. vaginatum*, with the other recorded species only appearing in a few quadrats and then not all of the time. Therefore, we restricted our attention to the leaf statistics from *E. vaginatum*. In addition, leaf width in *E. vaginatum* was found not to vary (it was almost always 0.5 mm or less) so we considered leaf length as the main indicator of vigour. As well as graphing the leaf length data, we investigated the effects of treatment, quadrat position (east or west) and year using the same ANOVA as applied to the nested frequency data.

We separated position (east or west of the ditch) in the vegetation analysis because of uncertainty about which side might show the greater drainage effect in two of the control ditches (Holden et al. 2017). After ditch blocking, water could flow across some blocked ditches towards the control (open ditches). Because the orientation of the ditches was somewhat oblique to the steepest gradient (see Fig. 1), this flow of water meant that there was a possibility the eastern sides of two of the open ditches could become wetter after blocking took place and not act as reliable controls.

To account for differences in the intervals between manual measurements of water-table depth, we used time-weighted annual means in our statistical analysis. This removes biases that may be caused by a higher frequency of readings at one time of year compared to another. We analysed separately how the weighted average annual water tables 2 m to the west and 2 m east of the ditches varied between the treatments and between years (1, 2, 3 and 4: 2011–2014) using a repeated-measures ANOVA (in IBM SPSS Statistics version 23). In a similar fashion we considered the effect of treatment and year on average summer water-table depths. We did so because summer water tables may have a greater effect on vegetation composition than water tables at other times of year. We defined summer water table as the time-weighted

average water-table depth for the period spanning June, July and August.

Results

Permanent quadrats: abundance of sedges and *Sphagnum* spp.

Summary data from the 24 quadrats have been separated according to plant grouping (sedge and *Sphagnum*) and whether the quadrat was to the east or west of the ditch and are shown in Tables 1, 2, 3 and 4. The data in the tables suggest the following:

- For the quadrats to the east of the ditches:
 - Sedge: sedges were very abundant across all of the treatments during the period of the experiment. There is the suggestion of an increase in abundance in Year 4 (2014), with differences in abundance between the other years being less obvious.
 - *Sphagnum*: in general, there were year-on-year increases in nested frequency of *Sphagnum* spp. in the control and dammed treatments, but no obvious pattern over time for the reprofiled treatment. The initial difference in average *Sphagnum* abundance between the control and dammed remained over time as abundance in both increased broadly in parallel.
- For the quadrats to the west of the ditches:
 - Sedge: the nested frequency of sedges was consistently high across all years in the control and dammed treatments. The reprofiled treatment had somewhat lower nested frequencies between Year 1 (2011) and Year 3 (2013) than the other treatments.
 - *Sphagnum*: the nested frequency of *Sphagnum* spp. increased in all treatments between Year 0 (2010) and Year 5 (2015). *Sphagnum* abundance was consistently higher in the control (open ditch) treatment than in the two re-wetting treatments. Both re-wetting treatments had a similar level of *Sphagnum* abundance.

The ANOVA test on the sedge abundance data revealed that there was no treatment effect ($p = 0.903$) or position (east, west) effect ($p = 0.983$). Time (year

Table 1 Nested frequency (%) of sedges (mainly *Eriophorum vaginatum*) from the permanent quadrats 2 m to the east of the ditches (n = 4)

	Year 0	Year 1	Year 2	Year 3	Year 4	Year 5
Control						
Mean	94.3	92.5	82.0	87.5	95.3	90.3
Median	94.0	95.0	86.5	93.0	97.0	91.0
Max	100	100	100	100	100	100
Min	89	80	55	64	87	79
Dammed						
Mean	70.8	84.5	92.3	86.8	96.3	82.5
Median	91.5	88.5	91.5	92.0	100.0	85.5
Max	100	98	100	100	100	99
Min	0	63	86	63	85	60
Reprofiled						
Mean	94.3	91.0	81.8	89.5	96.5	96.8
Median	96.5	94.5	84.0	94.0	99.5	100.0
Max	100	99	97	98	100	100
Min	84	76	62	72	87	87

Table 2 Nested frequency (%) of *Sphagnum* spp. from the permanent quadrats 2 m to the east of the ditches (n = 4)

	Year 0	Year 1	Year 2	Year 3	Year 4	Year 5
Control						
Mean	39.8	46.0	49.8	51.5	66.5	70.8
Median	38.5	41.0	50.0	57.0	74.0	77.5
Max	70	83	79	92	90	96
Min	12	19	20	0	28	32
Dammed						
Mean	15.3	18.5	24.3	30.3	35.5	29.8
Median	10.0	23.0	28.0	34.5	33.5	28.5
Max	41	27	35	48	65	50
Min	0	1	6	4	10	12
Reprofiled						
Mean	30.5	19.5	22.5	27.3	33.0	30.5
Median	23.5	19.0	23.5	20.0	36.5	33.5
Max	60	39	43	67	59	55
Min	15	1	0	2	0	0

of measurement) has a significant influence ($p = 0.037$) with Year 4 = Year 5 > Year 1 = Year 2 (Year 0 and Year 3 were not significantly different from any other year). There is no interaction effect between time, treatment and position ($p = 0.867$). The analysis of the *Sphagnum* nested frequency data revealed no effect of treatment ($p = 0.088$) or position ($p = 0.298$). Time has a significant influence ($p < 0.0001$), with Year 5 = Year 4 > Year 3 > Year 2 = Year 1 > Year 0. There is no interaction effect between time, position and treatment

($p = 0.952$). In short, these results suggest (a) that drainage/blocking treatment has had no effect on sedge and *Sphagnum* abundance and (b) that significant inter-annual changes in sedge and *Sphagnum* abundance can occur.

Vigour (leaf length) of *E. vaginatum* in the quadrats

The leaf length (vigour) data from the 24 quadrats 2 m either side of each ditch are shown in Figs. 3 and 4.

Table 3 Nested frequency (%) of sedges (mainly *Eriophorum vaginatum*) from the permanent quadrats 2 m to the west of the ditches (n = 4)

	Year 0	Year 1	Year 2	Year 3	Year 4	Year 5
Control						
Mean	92.0	90.3	84.3	95.3	92.8	95.0
Median	93.5	97.0	90.5	95.5	94.5	94.5
Max	100	98	98	96	100	100
Min	81	69	58	94	82	91
Dammed						
Mean	91.8	90.3	93.8	93.8	95.3	95.0
Median	91.0	90.5	93.0	97.0	98.0	96.0
Max	97	100	99	98	100	97
Min	88	80	90	83	85	91
Reprofiled						
Mean	94.3	63.0	79.0	85.8	86.5	89.8
Median	92.5	61.0	82.0	86.0	100.0	100.0
Max	100	96	100	100	100	100
Min	92	34	52	71	46	59

Table 4 Nested frequency (%) of *Sphagnum* spp. from the permanent quadrats 2 m to the west of the ditches (n = 4)

	Year 0	Year 1	Year 2	Year 3	Year 4	Year 5
Control						
Mean	18.0	36.5	34.8	38.3	49.5	51.0
Median	21.0	40.0	34.0	49.5	53.0	52.0
Max	28	66	71	54	92	100
Min	2	0	0	0	0	0
Dammed						
Mean	4.8	14.5	16.3	23.8	32.5	35.5
Median	2.5	3.5	7.5	9.0	26.0	39.5
Max	13	51	46	73	57	53
Min	1	0	4	4	21	10
Reprofiled						
Mean	8.5	8.8	9.8	22.0	26.5	33.0
Median	5.0	3.0	3.5	7.5	4.5	16.0
Max	24	29	32	73	97	100
Min	0	0	0	0	0	0

The graphs suggest that leaf length did not vary with treatment or position, but that there is a possible trend of decreasing leaf length over time. The statistical analysis confirms these visual impressions. The repeated-measures ANOVA revealed no treatment ($p = 0.945$) or position ($p = 0.585$) effects. Time (year of survey) did have a significant influence ($p < 0.0001$), with Year 0 = Year 1 = Year 2 > Year 5 = Year 3 = Year 4. There was no interaction effect between time, position and treatment ($p = 0.353$).

Sphagnum abundance increases over time, while sedge vigour is lower in the latter part of the study period which may suggest a relationship or association between the two. We tested this possibility using repeated-measures mixed models in either direction (i.e., *Sphagnum* abundance and then sedge vigour as the dependent variable). In each case, the vegetation-related independent variable (*Sphagnum* abundance or sedge vigour) was removed from the model, with year (time) being the dominating driver of variance.

Fig. 3 Mean leaf length of *Eriophorum vaginatum* in the quadrats 2 m to the west of the ditches. Error bars show ± 1 SE

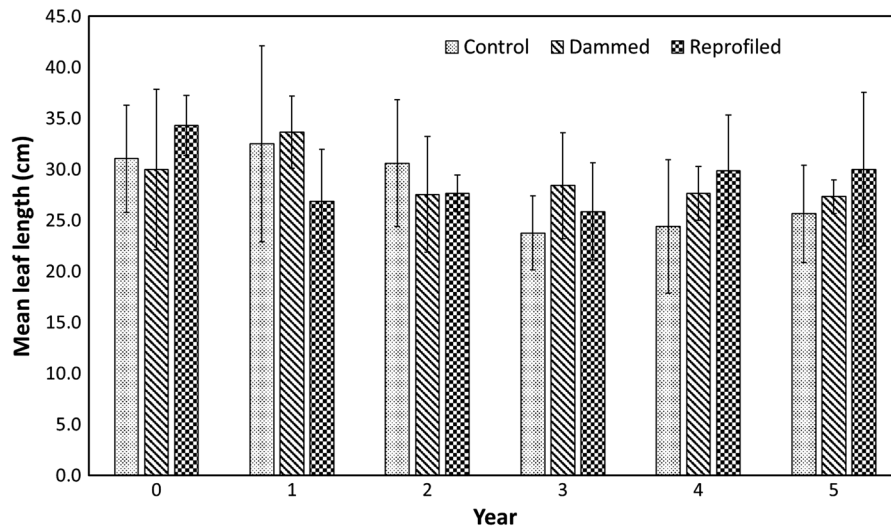
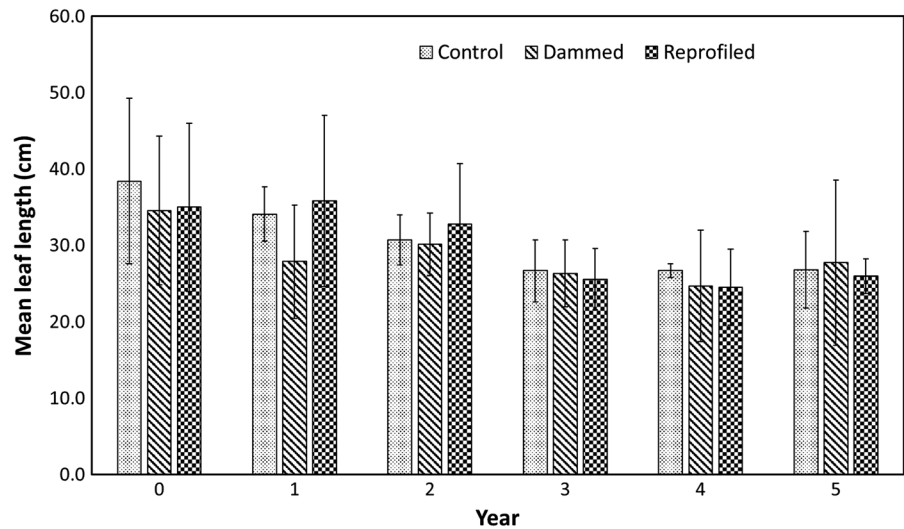


Fig. 4 Mean leaf length of *Eriophorum vaginatum* in the quadrats 2 m to the east of the ditches. Error bars show ± 1 SE

Therefore, there does not appear to be an association between the two.

Water-table depths

Water tables at the study site tended to be shallow—mostly within 15–20 cm of the surface. Table 5 summarises the weighted annual average water table depths for the dipwells 2 m east and west of the ditches. The dammed and reprofiled treatments to the east of the ditches had a weighted annual average water-table depth of 8.1 and 8.4 cm, respectively, whilst for the drained treatment (control) the depth

was 11.2 cm (Years 1–4). For the dipwells west of the ditches the dammed and reprofiled treatments had a weighted annual average water-table depth of 8.5 and 8.4 cm, respectively, whilst that for the drained treatment (control) was 13.4 cm (Years 1–4). Nevertheless, for both sets of dipwells there was no significant effect of drainage treatment (west $p = 0.211$; east $p = 0.733$). In both cases there was an effect of year (west $p = 0.001$; east $p = 0.018$). For the west-of-ditch dipwells, water tables in Year 3 were significantly deeper (by *c.* 1.7 cm) than in Year 2, while for the east-of-ditch dipwells water tables in Year 4 were significantly deeper (by *c.* 1.1 cm) than in

Table 5 Weighted annual average water-table depths for the dipwells 2 m east and west of the ditches (cm) (n = 4)

	Annual average weighted water table depth (cm)			
	Year 1	Year 2	Year 3	Year 4
Control				
East	11.0 (9.2)	10.3 (9.2)	11.4 (9.1)	12.2 (9.3)
West	13.6 (5.4)	12.5 (5.4)	13.9 (5.9)	13.8 (6.2)
Dammed				
East	7.8 (1.7)	7.6 (1.8)	8.5 (1.9)	8.5 (1.9)
West	8.2 (2.4)	7.6 (3.3)	9.7 (3.2)	8.6 (3.1)
Reprofiled				
East	8.1 (4.7)	8.1 (5.1)	8.8 (5.2)	8.5 (5.2)
West	8.2 (3.9)	7.7 (3.7)	9.4 (3.5)	8.4 (3.6)

Parentheses show ± 1 SE

Year 2. When the east and west dipwells are combined, the same trends are, unsurprisingly, apparent: there is no significant effect of drainage treatment ($p = 0.197$) on water tables, but a significant effect of year ($p < 0.0001$), with Years 3 and 4 having significantly deeper water tables (by *c.* 1.3 cm) than Year 2.

Table 6 summarises the weighted summer average water table depths for the dipwells 2 m east and west of the ditches. The dipwells east of the ditches had weighted average summer water-table depths of 12.4, 11.9, and 17.3 cm (dammed, reprofiled, and open (control) respectively; Years 1–4). The equivalent figures for the dipwells to the west were 12.1, 11.6, and 14.2 cm. There was no significant effect of drainage

Table 6 Weighted summer average water-table depths for the dipwells 2 m east and west of the ditches (cm) (n = 4)

	Summer average weighted water table depth (cm)			
	Year 1	Year 2	Year 3	Year 4
Control				
East	14.9 (6.4)	12.0 (4.8)	21.4 (6.9)	20.8 (6.4)
West	12.7 (9.9)	9.5 (8.4)	17.4 (9.5)	17.1 (9.7)
Dammed				
East	9.9 (2.5)	6.6 (3.2)	17.5 (2.8)	15.5 (3.3)
West	10.3 (2.4)	6.9 (2.0)	16.6 (3.3)	14.5 (2.3)
Reprofiled				
East	9.9 (3.8)	7.1 (3.6)	16.4 (3.4)	14.3 (3.5)
West	9.4 (3.6)	7.4 (5.4)	15.4 (4.8)	14.2 (5.6)

Parentheses show ± 1 SE

treatment for either east or west dipwells (west: $p = 0.209$; east $p = 0.826$). A significant effect of year on summer water tables was, however, apparent (west: $p < 0.0001$; east: $p < 0.0001$), with three distinct subsets: Year 3 (west 18.4 cm; east 16.5 cm) = Year 4 (west 16.9 cm; east 15.3 cm) > Year 1 (west 11.6 cm; east 10.8 cm) > Year 2 (west 8.6 cm; east 7.9 cm). When the east and west dipwells are combined there was no significant effect of drainage treatment ($p = 0.294$). A significant effect of year on the combined dataset was apparent ($p < 0.0001$), with three distinct subsets: Year 3 (17.4 cm) = Year 4 (16.1 cm) > Year 1 (11.2 cm) > Year 2 (8.3 cm).

Discussion and conclusion

Our data show no evidence that drain blocking leads to changes in sedge or *Sphagnum* abundance on blanket peatland. Similarly, there were no significant effects of drain blocking on annual average or summer average water tables. Given the water-table results it is not surprising that there is no vegetation effect. The lack of a treatment effect on water tables might seem somewhat surprising because the annual average and summer average water-table means for the dipwells next to the open ditches were consistently (i.e., across all four years) more than 2 cm deeper than for the wells next to the blocked ditches. However, the lack of a treatment effect may, in part, be a reflection of the relatively small sample size (12 wells in total and four per treatment) and high variability among the wells.

Our water table results contrast with those of Holden et al. (2011) and Wilson et al. (2010). Both of those studies found a significant, albeit small (2–3 cm), difference between water tables next to open ditches and blocked ditches. Several explanations may be advanced for why our study differs from these previous investigations. One possibility is sample size as noted above. Another possibility relates to climate. Our site in the Migneint appears to be considerably wetter than the sites used by Holden et al. (2011) and Wilson et al. (2010). As we note above (Materials and methods, Study site and meteorological conditions) we recorded 2255, 2409, 1786 and 1936 mm in each year of the water-table part of the study (Years 1–4). Oughtershaw Moss which was studied by Holden et al. (2011) has an annual average

rainfall of 1774 mm but was relatively dry during the period when water tables were monitored by the authors (2005 rainfall: 1567 mm, January to June, 2006 rainfall: 517 mm). Rainfall data for the individual sites studied by Wilson et al. (2010) are not available, although a lower-altitude gauge in the wider catchment has an annual average rainfall of 1635 mm (UK Met Office: goo.gl/QWQGeb; last accessed on 02.09.2016), and the years in which Wilson et al. (2010) took measurements appear to have been close to this average. The effect of a ditch on neighbouring water tables takes time to occur; water has to flow into the ditch to lower the adjacent water table. In wetter climates the soil between ditches will tend to be topped up more regularly with water so that the effect of a ditch on water tables is less clear-cut. Therefore, in wetter climates a drainage effect may be less obvious and more difficult to detect. A third explanation concerns the orientation of drains in the different studies. Our study drains more or less followed the gradient, while those investigated by Holden et al. (2011) and Wilson et al. (2010) were mostly cross-slope. Both types of drain configuration are common in the UK, but their effectiveness in controlling water tables may differ.

We have focused on water-table depths, because of their assumed ecological significance, rather than on how ditches affect the absolute position of the water table relative to a datum. However, changes (over time) to the peat adjacent to ditches may mask or complicate any effects ditches have on water-table depths, especially some years (>5–10 years) after ditch construction. Alterations to the peat alongside a ditch might be expected especially in the period immediately after ditch construction. As water tables fall next to the ditch they expose peat to rapid oxidic decay and, in response, the peat may subside so that the peatland surface adjacent to ditches is lower than that midway between them. These possible changes have yet to be studied properly, but fine-scale topographic data from our study site in the Migneint are consistent with such processes occurring: we found the ground-surface midway between ditches was higher than the peat surface 2 m either side of the ditches which, in turn, was higher than the peat surface immediately next to the ditch (Holden et al. 2016, 2017); in other words there was a clear surface slope perpendicular to the ditch (towards the ditch channel). The outcome of such changes in the peatland

surface is a reduction in the apparent drainage effect of a long-established ditch when considered in terms of water-table depths. Peat shrinkage brings the peat surface closer to the water level in the ditch, thus reducing water-table depths. Therefore, even though absolute water-table position (i.e. height above a datum) close to a ditch may be considerably lower (10 s of cm) than that midway between ditches, water-table depths may be similar between the two locations. Williamson et al. (2017) have termed this effect ‘self-rewetting’ and use it to explain a lack of effect of ditch blocking on blanket peatland vegetation. When ditches are blocked, the water level may be raised quite substantially in the ditch channel itself but this may have relatively little effect on water tables 2 m either side of the ditch where the sloping peatland surface is the greatest influence on the rate at which water is shed into the ditch and on water-table dynamics. Fine-scale topographic data for the areas between ditches are not reported by Holden et al. (2011) and Wilson et al. (2010) so it is not clear how much this effect may have influenced their data; the subsidence effect will depend on the initial size of the ditch and on the properties of the peat in which the ditch is constructed.

No clear temporal trends in sedge abundance were found, in contrast to the nested frequency of *Sphagnum* which increased fairly steadily over time. An apparent visual ‘correlation’ of this increase with a decrease in sedge vigour was not borne out in additional statistical analysis, and a change in sedge vigour cannot be used to help explain the *Sphagnum* trend (cf. Heijmans et al. (2002)). The hydrological analysis shows deeper water tables in Years 3 and 4, and these were the years during which *Sphagnum* abundance continued to increase, which is perhaps the opposite of what might be expected; it is usually assumed that wetter conditions will favour *Sphagnum* growth (e.g., Campeau and Rochefort 1996), although lags in plant response to water-table conditions may mean that it is not appropriate to compare plant abundance data directly with water-table data from the same year. Nevertheless, these results also suggest that a focus solely on water table as a control on peatland vegetation is unwise. A wide range of ecological and environmental factors can affect peatland plant growth. For example, Murray et al. (1993) investigated photo-inhibition in *Sphagna* in both field and laboratory experiments. *Sphagnum* plants exposed to

high-light conditions, even for short periods of time (48 h), showed reduced photosynthetic capacity for periods of at least two weeks. So, even in a generally wet year, a few short periods of intense solar radiation (bright days) might substantially inhibit *Sphagnum* growth for the rest of the year, thus confounding any relationship between abundance and average water tables. That the water table is not always a primary control on peatland plant distributions may explain the somewhat complicated picture that emerges in the studies by Stewart and Lance (1991) and Bellamy et al. (2012) as reviewed in the Introduction.

Finally, it might be argued that any treatment effects at our study site have not had long enough to be expressed in the vegetation and that a longer period of investigation may reveal responses to blocking. This is, of course, possible, and it is clear from other studies, including Stewart and Lance (1991), that changes in abundance can occur over many years (decades). However, we did find significant inter-annual changes in the two plant groups that we studied, showing that blanket bog vegetation can change substantially between years. And, although we found year-on-year changes in *Sphagnum* abundance, these were not limited to the drain blocking treatments.

End note

To arrive at this estimate, we used a blanket peatland area of 15,736 km², assumed a mean peat depth of 1.5 m, a mean dry bulk density of 100 kg m⁻³, and a mean carbon content of peat dry mass of 50%.

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