Knockfin Heights: a high-altitude Flow Country peatland showing extensive erosion of uncertain origin

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SUMMARY

Peatland erosion has important implications for ecosystem services such as carbon storage and biodiversity conservation. The extensive blanket bogs of Scotland's Flow Country include some of the least damaged peatlands in the UK; nevertheless, erosion features occur widely in the area. Here we describe a high-altitude (340–440 m) heavily eroded peatland at Knockfin Heights in the Flow Country. The causes of this erosion are poorly understood, and could include both human and natural factors. Restoration management has been considered for the site, although this might be unnecessary if spontaneous recovery is taking place. Our study aimed to contribute to understanding whether recovery was occurring, by (i) accurately and repeatably measuring bare peat extent using ground surveys, in 2005; and (ii) comparing the fullness of pools (strictly, pool area covered in standing water) using aerial photographs taken during 1946–2016. Estimated bare peat cover in the study area in 2005 was 10.5 % (95 % confidence limits 6.8–15). When placing a horizontal 1 m cane on the ground, bare peat was recorded under an average of 2.7 (2.2–3.2) 10 cm sections out of a possible maximum of ten per cane. Bare peat was most extensive on the flattest ground. Pools largely covered by standing water made up 30–60 % of pools in five pool groups, in five surveys during 1946–2005; but 75 % (standard error ~ 8) in 2016 - significantly higher than some earlier years. Active restoration management is merited for anthropogenic erosion but is more questionable for erosion that arose naturally or shows strong spontaneous recovery. We recommend follow-up studies during the next ten years to clarify the trajectories and causes of erosion at Knockfin; specifically, a repeat of this ground survey alongside remote sensing, experimental and palaeoecological work.

KEY WORDS: aerial photographs, bare peat, blanket bog, deer, pools

INTRODUCTION

While peatlands can show remarkable long-term stability, they are sensitive to certain thresholds (e.g. in hydrology or management) which, if crossed, can lead to extensive erosion (Bragg & Tallis 2001). The subject of peatland erosion has attracted considerable interest (Evans & Warburton 2010), reflecting its potential impact on important ecosystem services such as carbon storage (Evans *et al.* 2015), biodiversity conservation (Littlewood *et al.* 2010) and water quality (Wilson *et al.* 2011). Peatland erosion is particularly widespread in the UK, where 16 % of peatland is considered eroded (Tallis 1998). The most severe erosion tends to occur at higher altitudes across broad watershed plateaux (Lindsay 2010) and at exposed, northern, high-rainfall locations (Cummins *et al.* 2011). In some areas, erosion is largely attributed to human impacts such as accidental wildfire (Mackay & Tallis 1996) or atmospheric pollution (Woodin & Farmer 1993). Such erosion would merit active restoration management, especially in the absence of strong spontaneous recovery (Minayeva *et al.* 2017). Restoration management has been shown to increase vegetation cover and raise the water table; although improved monitoring, evaluation and information-sharing are needed (Parry *et al.* 2014, Andersen *et al.* 2016).

However, the causes of peatland erosion are often difficult to determine. Multiple factors, both natural and human, may have influenced a present-day eroded landscape over different temporal and spatial scales (Evans & Warburton 2010). Some wholly natural processes might trigger erosion, for example when climatic changes and/or the instability of peat soils result in a threshold being crossed, whereby forces that can promote erosion (such as wind or water movement) exceed the capacity of the vegetation and other attributes of the mire system to maintain stability (Bragg & Tallis 2001). Such naturally-eroding peatland features and landscapes would contribute to the diversity of landforms that adds so much interest to natural peatland ecosystems

(Minayeva *et al.* 2017). Some well-known bare peat features are clearly of natural origin; e.g., permafrost collapse scars (Camill 1999). The ability of peatland plants to rapidly colonise bare peat (reviewed by Evans & Warburton 2010) and the selection of bare peat by some peatland birds for foraging (Pearce-Higgins & Yalden 2004, Massey *et al.* 2016) suggest that some species may be pre-adapted by evolution to exploit exposed peat within natural peatlands.

The Flow Country of northern Scotland is a ~ 4000 km^2 peatland landscape characterised by blanket bogs, pool complexes and lakes (Lindsay *et al.* 1988). Despite past impacts, the most serious being widespread afforestation with non-native conifers in the 1980s (Stroud *et al.* 1987), remaining areas of the Flow Country hold some of the least damaged peatlands in Britain (Littlewood *et al.* 2010) in terms of anthropogenic impacts like drainage, cutting, fires, over-grazing and pollution. The nature conservation importance of the area is reflected in the designation of over 1300 km^2 as Natura 2000 sites under the European Habitats and Birds Directives, including the largest terrestrial Special Area of Conservation (SAC) in the UK. Its size, relatively intact state and deep peat soils (with large areas over 2 m deep) make the Flow Country a key area for soil carbon storage (Chapman *et al.* 2009). Erosion features are common but patchily distributed in the Flow Country; if these were natural in origin, they would be worthy of conservation (Lindsay *et al.* 1988).

Most Flow Country peatlands occur within seven bioclimatic zones (Lindsay *et al.* 1988); these include perhaps all the best-known relatively low-lying areas that are quite easily viewed and accessed from roads or tracks. Typically, peatlands in these seven zones have annual accumulated temperature and frost degree-days of 825–1100 and 20–110 day°C respectively, and mean wind speeds of 4.4–6.2 m s^{-1}. There are, however, a few Flow Country peatlands that occupy a cooler, windier bioclimatic zone (O2.H1/A3; Lindsay *et al.* 1988). These occur on gentle slopes above 330 m altitude, cannot be overlooked from roads or tracks, and a long walk in is required for any ground survey visits. The different climate in this zone, compared to elsewhere in the Flow Country, is illustrated by typical annual accumulated temperature and frost degree days of 550–825 and 110–230 day°C, respectively, and mean wind speeds of 6.2–8.0 m s^{-1} (O2.H1/A3; Lindsay *et al.* 1988). In this study we investigated the largest of these high-altitude Flow Country peatlands, namely Knockfin Heights. Perhaps paradoxically, Knockfin Heights was assessed in a nature conservation audit as one of the 'poorest' areas of the Flow Country for habitat quality indicators such as bare peat extent and *Sphagnum* cover, but also as one of the 'best' areas, for lack of apparent anthropogenic damage or threat (Matheson *et al*. 2003). Furthermore, its blanket bog condition is currently assessed as 'favourable' in statutory nature conservation site condition monitoring (http://gateway.snh.gov.uk/sitelink). In general, eroding peatlands are under-represented among nature conservation sites in Scotland, probably because they are often viewed as being in poor habitat condition (Cummins *et al.* 2011).

The impetus for our study followed the 1995 establishment of the nature reserve at Forsinard Flows in the heart of the Flow Country. The reserve has the prime aim of maintaining and enhancing its peatland habitats and associated fauna, including the wading birds (Charadrii) for which the Flow Country is of particular importance (Stroud *et al.* 1987). The reserve includes part of the Knockfin Heights plateau, an area that seemed (paradoxically) rich in wading birds but poor in apparent habitat condition, with multiple empty pools and signs of extensive erosion. This prompted a discussion as to whether restoration management should be considered. However, there was little evidence to clarify whether the bare peat features were naturally or anthropogenically caused. Lacking the resources to carry out detailed research into the causes, we focused on a more tractable question: was the area of bare peat increasing or decreasing? Knowing whether there was an increase or decrease in bare peat area would not enable us to identify past causes of erosion, but it would help us to understand the potential future trajectory for the site. This, in turn, would help to inform management decisions. In particular, if spontaneous recovery is taking place, then there may be no need for active restoration management. We also wished to quantify the water levels of the pools and assess whether they might have changed in recent decades, using aerial photographs, to better understand how erosion and pool complexes were related - an often-neglected question (Evans & Warburton 2010). We considered it plausible that there were links between the two; for example, that some eroded areas might have begun as empty pool systems, or that erosion in surrounding areas might have caused some pools to empty. We recognised that determining whether such links were present would be beyond the current study. Nevertheless, evidence that bare peat was re-vegetating and more pools were holding water over time would suggest a degree of spontaneous recovery which, in turn, would have management implications, as set out above. Our aims were (i) to establish baseline field-survey measures of peatland condition which could be repeated accurately after many years, allowing changes to be measured even if they were slow; and (ii) to measure how many pools were full of water, using field survey combined with aerial photographs from 1946–2016, and see if this changed over time. Our aims in seeking to publish this study here are to bring this work (hitherto described only in an internal report) to a wider audience with a peatland interest and range of specialisms. We hope this will facilitate follow-up research to elucidate the origin and trajectory of such eroded areas. Thus, our aims reflect increasing calls for wider sharing of long-term data on peatlands, to foster improved and longer-term monitoring, which will ultimately support better management and policy decisions (Parry *et al*. 2014, Andersen *et al*. 2016).

METHODS

Study site

The study site was the Knockfin Heights plateau in the Flow Country of northern Scotland (Figure 1). We studied the higher, flatter part of the plateau, above 340 m a.s.l., which extends to about 56 km^2 and up to a maximum altitude of 438 m a.s.l. (58° 17.7' N, 3° 50.1' W). The study area comprises parts of two Sites of Special Scientific Interest (SSSIs) protected under UK nature conservation law since 1992, namely Knockfin Heights and Rumsdale Peatlands. These form part of the Flow Country Natura 2000 sites (protected under EU legislation), the Caithness & Sutherland Peatlands Special Protection Area (SPA; designated 1999) and Special

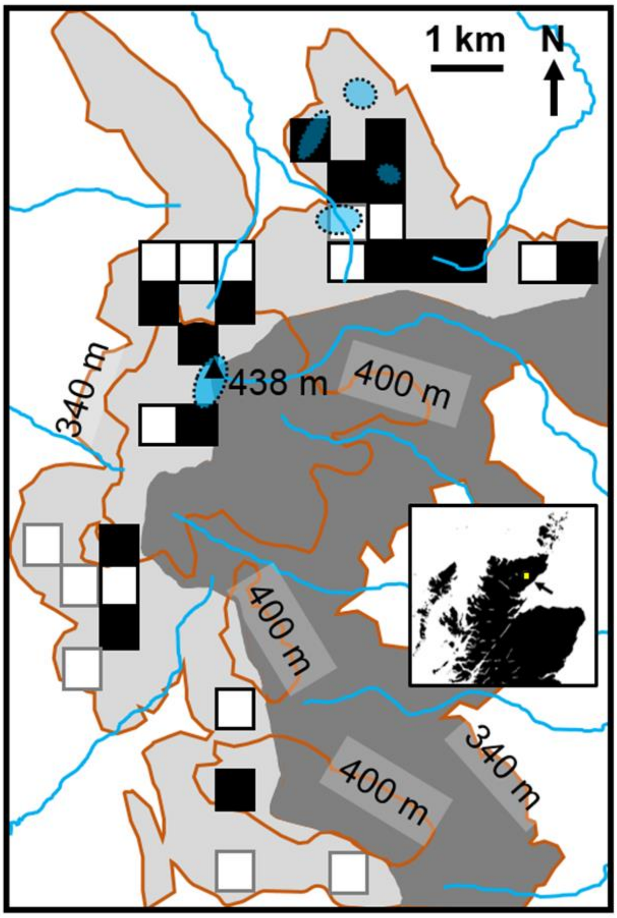

Figure 1. The study area: the Knockfin Heights plateau. The focal area was the part of the plateau above 340 m altitude, including the area which was sampled (light grey) and the area where permission to survey was not given (dark grey). The 340 m and 400 m contours are shown as dark orange lines. Survey squares are indicated in black (more eroded stratum) and white (less eroded stratum); a grey boundary indicates that the square was intended for survey but not completed (less eroded stratum only). The studied pool groups are marked as ellipses (light blue, semi-transparent, with dotted boundaries). The summit of the plateau (438 m) is shown as a small black triangle (location: 58° 17.7' N, 3° 50.1' W). The main streams flowing down from the plateau are shown as light blue lines. An inset map shows the location of the study area (yellow rectangle, arrowed) within northern Scotland.

Area of Conservation (SAC; designated 2005). The study site comprises parts of five private estates and one nature reserve owned by the UK nature conservation charity, the Royal Society for the Protection of Birds (RSPB). Permission to survey was granted for four of these landholdings, giving us access to 29 km^2 for field surveys.

Knockfin Heights is the largest contiguous high-altitude peatland in the Flow Country (Matheson *et al.* 2003). Its altitude probably makes its climate significantly cooler and wetter than that of the nearest weather station (Kinbrace; 58° 14' N, 3° 55' W, 103 m a.s.l.), where the 1981–2010 mean summer and winter temperatures were 10.7 °C and 3.9 °C, respectively, and annual rainfall 971 mm. Regional wind directions are predominantly from the south and west quadrants (180–300 °, www.metoffice.gov.uk).

The habitats of the study area largely comprise blanket mire and wet heath, characterised by plants such as the dwarf shrubs *Erica tetralix* L. and *Calluna vulgaris* (L.) Hull (Ericaceae), the graminoids *Eriophorum vaginatum* L. and *Trichophorum cespitosum* (L.) Hartm. (Cyperaceae), and the mosses *Sphagnum papillosum* Lindb. and *Sphagnum capillifolium* (Ehrh.) Hedw. (Sphagnaceae). Pools and small lakes are abundant. Wading birds such as Golden Plover *Pluvialis apricaria* (Linnaeus, 1758) and Dunlin *Calidris alpina* (Linnaeus, 1758) breed at relatively high densities. In Flow Country surveys in 2000 (Hancock *et al.* 2009), peak counts per 12.5 km transect at four plots on Knockfin Heights averaged 19.5 (standard error (s.e.) 6.7) and 5.5 (s.e. 2.1) for these two species, respectively, or roughly twice the means for 30 other plots (8.2 (s.e. 1.4) and 2.3 (s.e. 0.80), respectively) (unpublished data). The only large herbivores present are native wild Red Deer *Cervus elaphus* Linnaeus, 1758. On RSPB land within Knockfin Heights (989 ha), thrice-yearly deer counts during 2000–2003 gave a mean density of 4.6 (s.e. 2.3) animals per km^2 (unpublished data).

Study design

Our interest focused on the flatter parts of the plateau because these (i) appeared to hold the largest areas of bare peat, and (ii) were likely to be of most value to wading birds (Stroud *et al.* 1987): flat areas tending to hold more Golden Plover (Pearce-Higgins & Grant 2006) and abundant pools, which are important for wading birds such as Dunlin and Greenshank *Tringa nebularia* (Gunnerus, 1767) (Lavers & Haines-Young 1996, Hancock *et al.* 2009).

To identify areas for field survey in August 2005, we first divided the potential 29 km^2 study area into 500 m × 500 m grid squares. We estimated slope from the number of 10 m contours crossed by the two diagonals of each square (Stroud *et al.* 1987). We excluded more steeply sloping squares having, on average, more than five such contours per diagonal (which equates to a slope greater than approximately 4 °). We wished to ensure that our (randomly selected) survey squares included the full range of degrees of erosion present. Therefore, we used a brief walk-over survey and aerial photographs to divide the remaining area into two strata of approximately equal size, according to their (subjectively assessed) apparent erosion or bare peat extent. These 'more eroded' and 'less eroded' strata held 30 and 33 grid squares, respectively, from which we randomly selected 15 per stratum for survey. Due to the work taking longer than expected, and closure of some areas for deer stalking later in the season, we were unable to complete six of these squares, resulting in a surveyed sample of 15 'more eroded' and 9 'less eroded' squares. Within each survey square we established one 100 m survey transect at random, as follows. We overlaid a 100 m × 100 m grid, to identify the 16 grid intersections that were at least 100 m from the square perimeter. One of these intersections was selected at random as the start point for the transect, which was then established along a randomly selected bearing (see Appendix).

Field measurement of ground cover

Ground cover along survey transects was measured by (i) a large-scale survey, measuring linear cover in broad categories; and (ii) a fine-scale survey, measuring frequency for a range of plant species and other features. To establish an accurately repeatable baseline, we needed a method showing little between-observer variation that was capable of measuring quite small changes in cover. We expected that changes in bare peat cover might be quite slow, as found by Cummins *et al.* (2011).

To permanently set out each survey transect, we located the start point with a GPS handset (Garmin Etrex, reporting an estimated position error of 3–5 m) and marked it with a wooden post (1.5 m, 75 mm × 75 mm section). We made up a 50 m steel surveying cable, marked at 1 m intervals, and attached at each end to a pole. We used this cable, a hand-held compass and sighting along the transect to locate the mid- and end-points of the 100 m transect, which were then also marked with wooden posts. Positions on the ground below the cable were determined using a ranging pole fitted with a spirit level so that it could be held vertically.

To carry out the large-scale survey of (linear) ground cover, the 100 m transect was marked by setting up the 50 m cable first between the start and middle posts, then between the middle and end posts.

The observer walked slowly along the transect, identifying where the ground cover directly under the cable changed between five different categories: bare peat, vegetation, exposed rock, exposed mineral deposits, and open water. The cover category was considered to have changed when a different category was present over a length greater than 1 m (0.5 m for bare peat). Where smaller lengths of two categories occurred mixed together, we recorded this as (e.g.) "bare peat / vegetation mosaic", with the more abundant category indicated first. Where smaller lengths of three or more categories occurred together, we recorded this as 'mosaic'. We recorded (to the nearest cm) the distance from the transect start point to each boundary between cover categories, allowing us to calculate the linear cover of each category along each transect.

For our fine-scale surveys of ground cover, we initially considered using visual estimation of plant cover. However, this approach typically shows high levels of variation between observers (Sykes *et al.* 1983, Kennedy & Addison 1987). Therefore, we used a frequency-based method; such methods tending to show less observer variation (Kercher *et al.* 2003, Vittoz & Guisan 2007). Frequency measures were made at ten sample points per 100 m line transect, each located at the centre of a 10 m transect section. At each sample point, a 1 m measuring cane was placed on the ground, at right angles to the transect, with its centre directly below the cable. The cane was a 12 mm diameter wooden rod marked with ten 10-cm sections. Due to time constraints, we did not perform exhaustive botanical surveys, but instead focused on recording a few features that were considered particularly important measures of bog condition (Table 1). These features included various plant species, plant groups, and physical features such as bare peat. We measured the frequency of each feature at each sample point as the number of 10-cm sections per 1 m cane that lay over or under the feature in question. At each sample point, we took a digital photograph of the cane and probed the peat using a 150 cm rod (12 mm diameter). This allowed us to measure peat depths up to 140 cm; greater depths were recorded as '>140 cm'.

Characterising pools using ground surveys and aerial photographs

Pool groups were characterised by ground survey in August 2005, and by examining aerial photographs from five other years during the period 1946–2016.

Access restrictions in late August 2005 (due to deer stalking) meant that pool surveys were limited to the north end of the plateau, where we studied five pool groups (Figure 1). Groups were selected by first identifying a large pool as the start point, then visiting nearby pools over an increasing area until all pools in that group had been visited. An aerial photograph was carried in the field, and each pool was numbered on this photograph. The approximate percentage cover of the pool bed that comprised standing water, bare peat, vegetation or mineral deposits was estimated by eye. We photographed each pool, counted the number of channels connecting it to other pools, and estimated the orientation of its longest axis.

Later, we examined the same pools on aerial photographs from 1946 (August photograph for two pool groups, October for the remainder), 1980 (May), 1989 (May), 2001 (May) and 2016 (October). The images were enlarged for examination, and all the images used had sufficient resolution to assess the smallest pools in the sample (19 m^2). For each pool we estimated, from the aerial photograph, the percentage of the pool bed that was covered in standing water. Pool sizes were measured in the most recent photograph, including any parts of the pool bed that were not covered in water.

Data analysis

Means for the whole study area were estimated as the mean of the two stratum-level means, weighted by the total number of squares in each stratum. We used bootstrapping to estimate the 95 % confidence intervals of study-area means, as follows. In the more and less eroded strata, 15 and 9 squares, respectively, were surveyed. Therefore, we selected, at random, with replacement, 1000 bootstrap samples, each consisting of 15 and 9 data values (for the variable in question) from the surveyed squares in the more and less eroded strata respectively. From these, we calculated 1000 bootstrap estimates of the study-area mean, and selected the 25th smallest and 25th largest of these as the lower and upper 95 % confidence limits, respectively.

We also carried out two statistical analyses, using the GLIMMIX procedure in SAS (Stroup 2013). One analysis tested whether bare peat cover was correlated with slope, altitude, deep peat frequency or grazing indices. The other analysis tested whether the proportion of pools having the pool bed largely covered in standing water changed between years, after controlling for year-to-year variation in rainfall. To investigate correlates of bare peat cover, we collated the following data for each survey square ($N = 24$). The linear cover of bare peat was taken from the large-scale survey of ground cover (100 m line transects). We used 1:50,000 maps to estimate slope (in degrees, from the number of 10 m contours crossing grid square diagonals) and altitude (m above sea level at the square centre). Deep peat frequency was taken from fine-scale surveys of ground cover, and defined as the proportion of sample points on that 100 m transect having peat deeper than 1.4 m. Grazing of current growth was rarely recorded (only four of 24 squares) so we created a composite 'grazing index', as the mean of the frequencies of (i) grazing of current growth; and (ii) *Juncus squarrosus* L., a plant often considered to indicate heavy grazing in UK uplands (Table 1). Bare peat cover, square-root transformed, was then used as the *y*-variable in a general linear model with an identity link and normal error distribution. Slope, altitude, grazing index and deep peat frequencies were *x*-variables. We first fitted and examined the complete model, then simplified the model using backwards deletion in which *x*-variables having *P*-values > 0.05 were deleted in turn, starting with the variable having the highest *P*-value.

Table 1. Plants and other features recorded during fine-scale ground cover surveys. The reasons for selecting these features, and their interpretation, were based on expert advice on interpreting upland vegetation in the region, as set out in MacDonald *et al*. (1998) and further described in Lindsay *et al.* (1988) and Averis *et al.* (2004). Full scientific names including authority are given in the text, apart from *Pleurozia purpurea* Lindb., *Carex panicea* L., *Sphagnum tenellum* (Brid.) Bory and *S. magellanicum* Brid.

Category	Feature	Reason for selection / interpretation
Vascular plant species		
	Calluna vulgaris	Where abundant, indicates dry heath conditions
	Juncus squarrosus	Can indicate heavy grazing
	Carex panicea	Can indicate nutrient enrichment and/or peat erosion
Moss and liverwort species		
	Sphagnum capillifolium	Indication of presence of active blanket bog or wet heath conditions
	Sphagnum papillosum	Indication of presence of active blanket bog
	Sphagnum tenellum	Can indicate a change i.e. shift from bare peat to active bog or vice versa
	Sphagnum cuspidatum	Indication of presence of wet, active blanket bog / consistently high water table
	Sphagnum magellanicum	Indication of presence of undamaged, active blanket bog
	Racomitrium lanuginosum	Can indicate drier conditions, peat degradation, or recovery after a dry period
	Pleurozia purpurea	Indication of open habitat (associated with mire degradation in western Scotland)
Plant groups		
	All higher plants	Estimation of plant cover
	All *Sphagnum*	Estimation of *Sphagnum* cover and indication of presence of active blanket bog
	All other mosses	Indication of drier heath conditions
Other features		
	Grazing of current growth	Indication of deer grazing
	Bare peat	Estimation of extent of bare peat
	Bare rock/mineral	Estimation of extent of bare rock and mineral deposits
	Water over peat/mineral	Indication of high water table
	Water over vegetation	Indication of high water table

To test whether the proportion of pools holding water changed between years we first collated, for each pool group and year (five pool groups × six years, hence $N = 30$), the number of pools that had more than 50 % of the pool bed covered in standing water. We termed such pools ‘largely full’ although, strictly, we were measuring the area of pool bed covered in standing water rather than fullness *per se*. We wished to control for the fact that recent rainfall might affect how many pools were full, irrespective of between-year changes in the capacity of pools to hold water. Therefore, we collated rainfall data corresponding to the periods of our ground surveys and aerial photographs. Monthly rainfall records dating back to our earliest air photographs (1946) were available for the weather stations at Wick and Stornoway, around 40 km east and 100 km west of the study area, respectively. We averaged the figures from these two stations for each month to obtain an index of approximate monthly rainfall for our study area. We then collated these monthly figures for each survey month and each of the previous three months. Finally, we created four alternative measures of ‘recent rainfall’, as the rainfall index for the survey month or this index plus one, two or three previous months. We then fitted a generalised linear model with a logit link and binomial error distribution, using the number of pools that were ‘largely full’ as the *y*-variable and the number of pools in the pool group as the binomial denominator, so as to model the proportion of pools that were ‘largely full’ in each pool group and year. ‘Pool-group’ was fitted as a random effect, to model correlation between values for the same pool group in different years. We fitted ‘year’ as a categorical *x*-variable, and tested the effect of each of the four ‘recent rainfall’ measures in the model in turn. The rainfall variable having the strongest relationship with pool fullness (most positive parameter estimate) was retained in the final model, which was then used to estimate year-wise effects and to test for significant differences between all pairs of years using Tukey-adjusted *P*-values.

RESULTS

Ground cover of bare peat and vegetation

Extensive areas of bare peat were found during ground surveys, often as features such as hags (Figure 2A) and empty pools (Figure 2B). Large-scale surveys of ground cover along transects showed that bare peat averaged 13.6 % (s.e. 3.6) cover in the 'more eroded' stratum, around twice that of the 'less eroded' stratum (7.7 %; s.e. 2.8) (Figure 3). Across the whole study area, mean bare peat cover was estimated as 10.5 % (95 % confidence limits (c.l.s) 6.8–15). A further 41–43 % of ground cover (depending on stratum) comprised mosaic of bare peat and vegetation, with vegetation usually predominating (Figure 3). The longest single length of bare peat per 100 m transect averaged 6.1 m (s.e. 1.5) and 3.1 (s.e. 1.2) in the 'more eroded' and 'less eroded' strata respectively. Across the study area, the maximum length of bare peat per 100 m transect was estimated to average 4.5 m (95 % c.l.s 3.0–6.2). Peat depth was recorded as 'over 140 cm' at an average of 7.5 (s.e. 0.23) out of 10 points per 100 m transect; i.e., over an inferred 75 % of the area. Where peat was shallower than 140 cm, its depth averaged 99 cm (s.e. 30).

Fine-scale surveys of ground cover (Figure 4) showed that bare peat occurred at a mean frequency of 3.0 (s.e. 0.35) and 2.4 (s.e. 0.40) 10 cm sections per 1 m of cane in the more and less eroded strata, respectively. Across the whole study area, the mean bare peat frequency was estimated as 2.7 (95 % c.l.s 2.2–3.2). The frequency of combined *Sphagnum* species was low, averaging less than 1.0 in both strata. The most frequently recorded moss was *Racomitrium lanuginosum* (Hedw.) Brid., which was around three and six times more frequent than *Sphagnum* in the more and less eroded strata, respectively (Figure 4). Grazing of current growth was rare, as was *Juncus squarrosus*, which can indicate intense grazing (see Methods). *Calluna vulgaris* and *Sphagnum cuspidatum* Ehrh. ex Hofm. were somewhat more frequent in the less and more eroded strata, respectively. However, in general, there were few strong differences in fine-scale vegetation frequencies between the strata, contrasting with the clear between-strata difference in bare peat measures.

Statistical analysis of the correlates of bare peat cover showed that there was a highly significant ($P = 0.006$) association between bare peat and slope, with flatter areas having more bare peat (Table 2). The fitted relationship (Figure 5) implied that bare peat cover would rise from about 1.8 % on the steepest ground surveyed (slope: ~ 4 °) to around 20 % on flat ground; i.e. there was a roughly ten-fold increase in bare peat extent between the steepest and flattest areas surveyed. The other potential correlates (altitude, grazing frequency and deep peat frequency) showed no strong relationship with bare peat cover.

Pool characteristics and changes over time

Field surveys in 2005 covered 182 pools in five pool groups (Figure 1). Many pools had some of their bed exposed: on average, 57 % (s.e. 3.2) of potential pool area had no surface water, with exposed peat being

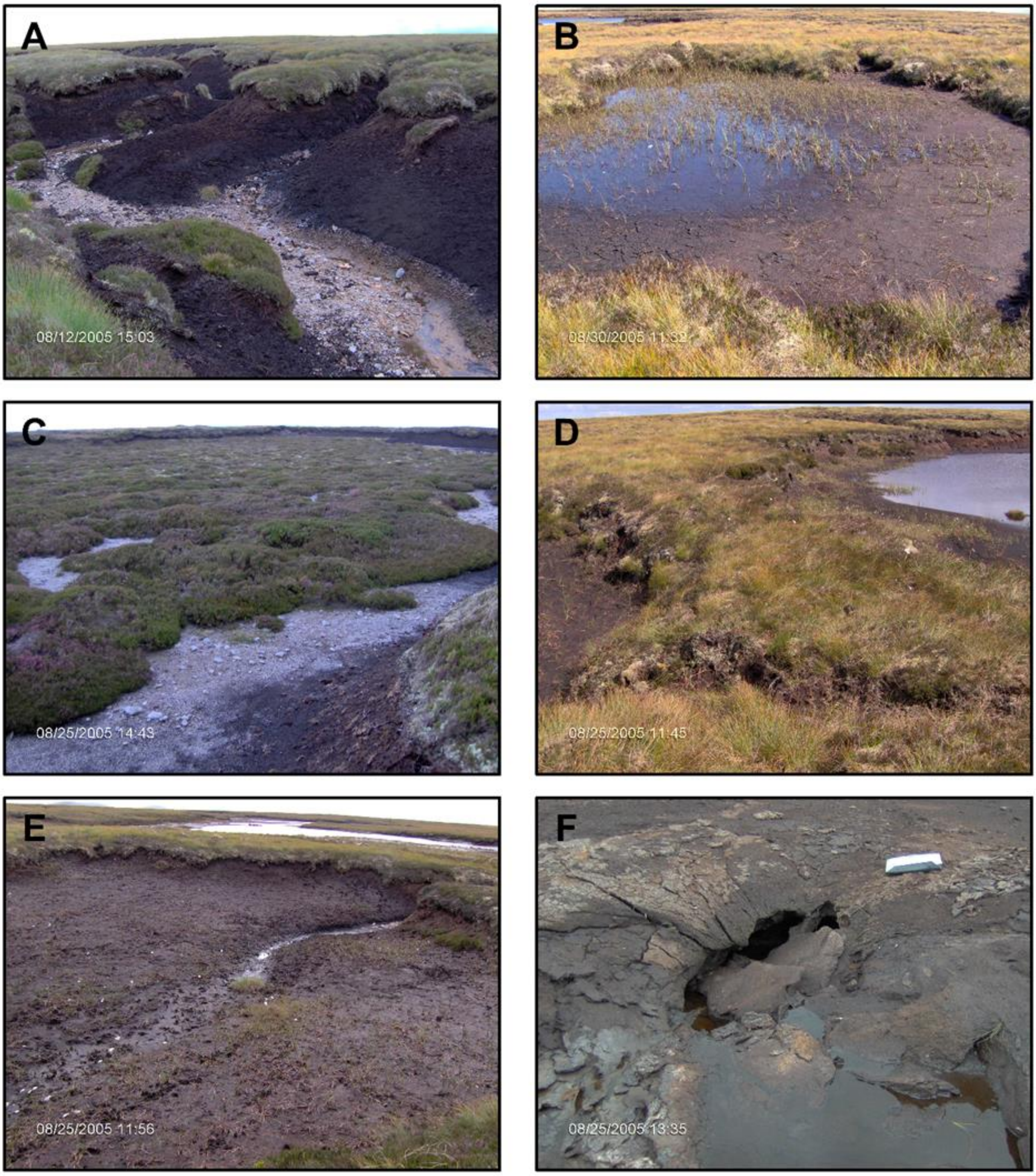

Figure 2. Photographs of different peatland features on the study site, taken during field surveys in August 2005. A: an example of a hagged area. B: a nearly empty pool, with extensive bare peat and patches of *Eriophorum angustifolium*. C: a pool apparently empty for many years, with some mineral soil and extensive *Calluna vulgaris*. D: An intact strip of land between two pools, showing the apparent development of a channel between the pools (foreground), marked by deer trampling. E: an empty pool, draining into a lower pool via a clear channel. F: a hole in the bed of an empty pool, apparently connecting to subterranean drainage such as a peat pipe.

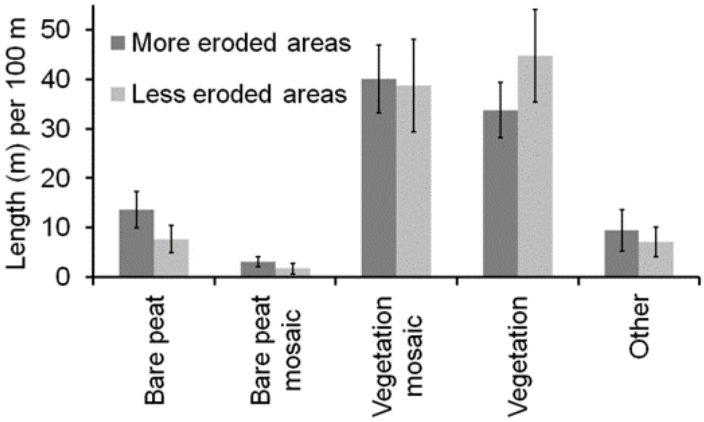

Figure 3. Large-scale ground cover in the study area: lengths of bare peat, vegetation and mosaics along 100 m transect surveys, by survey stratum (mean and s.e.). Mosaics are labelled according to whether bare peat, or vegetation, was the more common component of ground cover. 'Other' comprised open water, mosaics including open water, or more complex mosaics of three of more features.

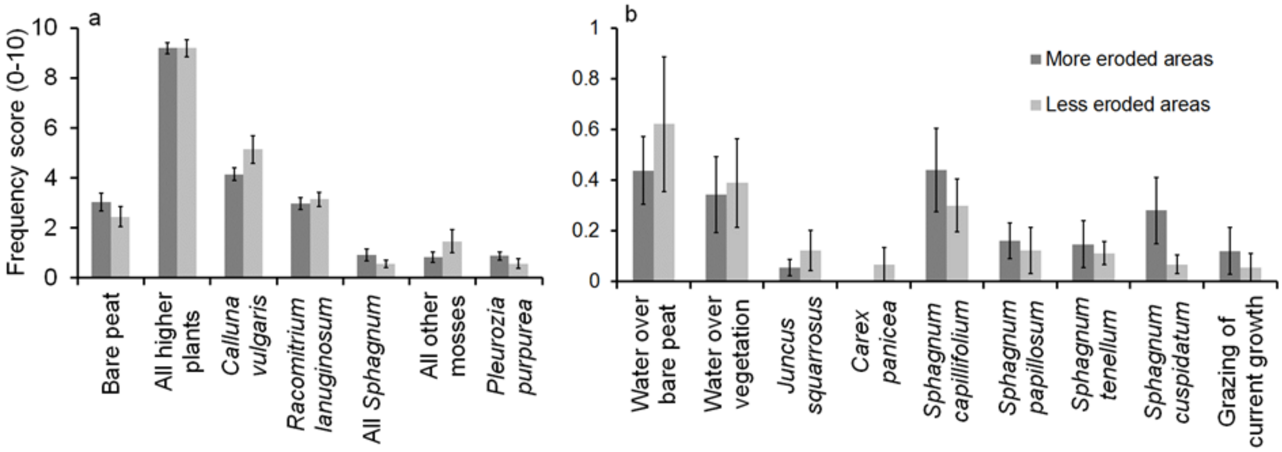

Figure 4. Fine-scale vegetation data from the study area: frequency scores of (a) common and (b) rarer plants and other features, by survey stratum (mean and s.e.).

Table 2. Statistical analysis of the correlates of bare peat cover. The *y*-variable was the linear cover of bare peat from 100 m line transects, square-root transformed. Unit of analysis: survey square ($N = 24$).

Model	Variable	Parameter estimate	Standard error	*P*-value
Full	Intercept	-2.95	4.31	0.502
	Deep peat frequency	0.65	0.98	0.517
	Grazing index	-1.70	2.09	0.425
	Slope	-0.75	0.31	0.025
	Altitude	0.02	0.01	0.110
Reduced	Intercept	4.62	0.62	< 0.001
	Slope	-0.89	0.29	0.006

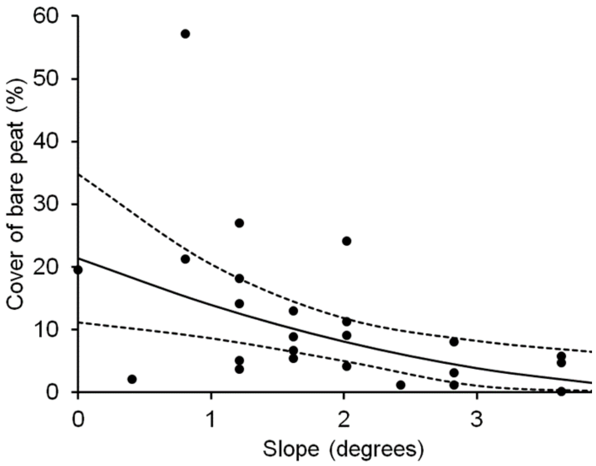

Figure 5. The relationship between slope and bare peat cover, as observed (black circles) and fitted (line, with 95 % confidence intervals). The fitted relationship is based on the reduced model (Table 2).

the most common substrate - making up 40 % (s.e. 2.7) of potential pool area (Figure 6a). Some exposed pool beds had patches of *Eriophorum angustifolium* Honck. (Figure 2B). Others had exposed mineral soil and cover of *Calluna vulgaris* (Figure 2C).

Most pools were connected by channels to other pools; the number of channels averaged 1.8 (s.e. 0.11) but ranged from zero to eleven (Figure 6b), suggesting a complex hydrology for the site. Pools without connections to other pools were often full (having over 90 % cover of standing water: 25 pools full out of 33, or 76 %). Meanwhile, pools that were connected to other pools were less often full (47 out of 149, or 32 %). Thus, the proportion of 'full' pools roughly doubled, when comparing pools with and without connections. The narrow strips or 'baulks' of land between adjacent pools sometimes appeared to have incipient channels forming (e.g. Figure 2D), often marked with deer hoof prints. In other cases,

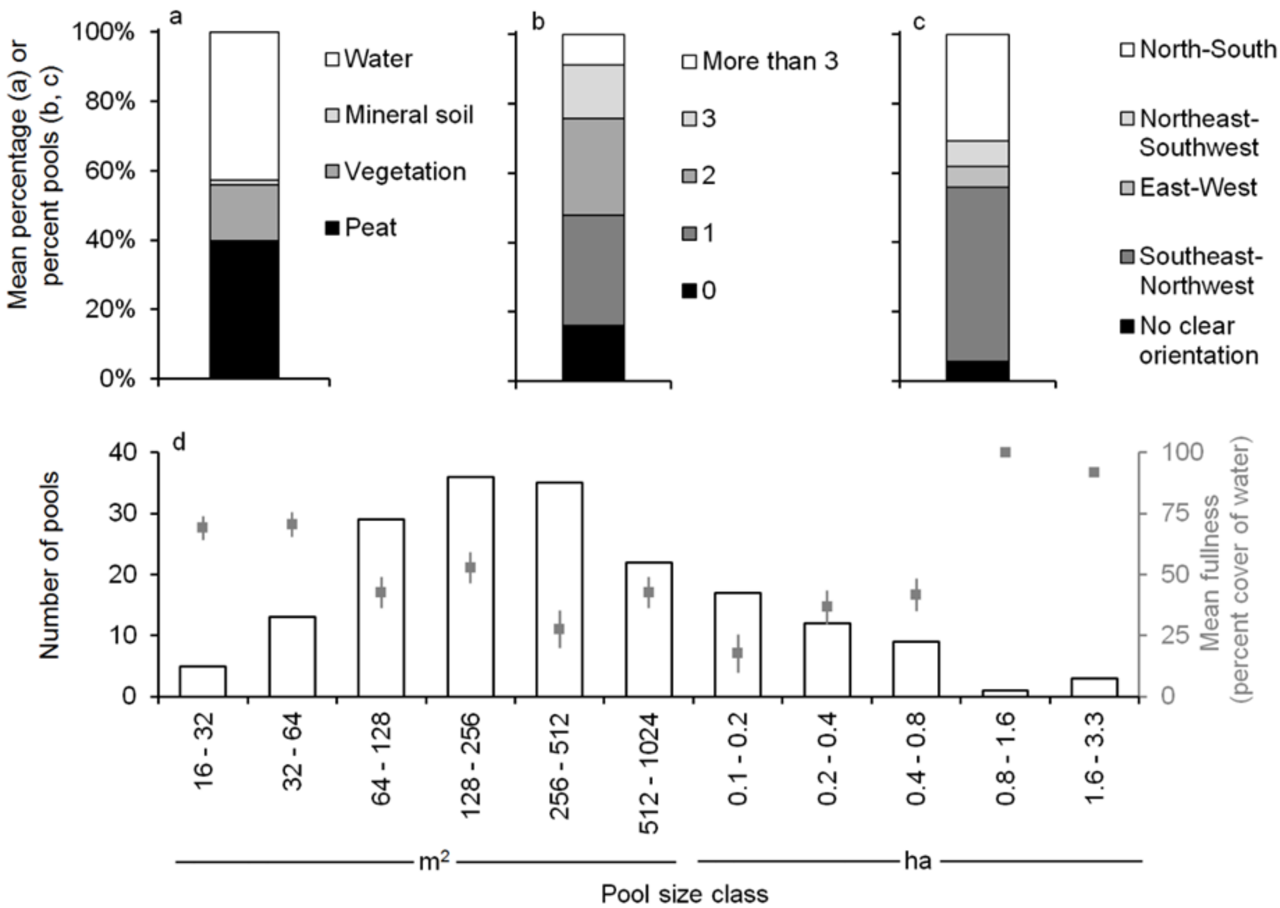

Figure 6. Pool characteristics in the five studied pool complexes: (a) cover of potential pool surface area (2005 field survey); (b) number of channels connecting with other pools (2005 field survey); (c) pool orientation; (d) pool size frequency distribution (white bars) (note that the *x*-axis is on a $\log_2$ scale); the secondary axis shows mean fullness (percent cover of water, 2005) by pool size class (grey squares, with standard errors).

empty pools were draining via a channel into an adjacent pool (e.g. Figure 2E). We observed holes in beds of empty pools, perhaps linked to subterranean drainage via a peat pipe or mineral soil (e.g. Figure 2F).

Most pools (145, or 80 %) had their long axis running northwest to southeast or north to south (Figure 6c). Pool sizes varied from 19 m^2 to 2.9 ha (median 300 m^2, mean 1260 m^2, s.e. 6.8) and showed a roughly log-normal distribution (Figure 6d). The larger pools tended to be less full (Figure 6d; $r_s = -0.17$, $N = 182$, $P = 0.024$, treating pools as independent). However, the four largest pools were full or nearly so (Figure 6d).

Analysis of pool 'fullness' (strictly, pool bed area covered by standing water) by year suggested that more pools were ‘largely full’ (had over 50 % cover of standing water) in 2016, than in three of the five earlier years (1946, 1989 and 2001) (Table 3; Figure 7). Even so, the proportion of 'largely full' pools was rather low: around 30–60 % in surveys between 1946 and 2005, and only reaching 75 % (s.e. ~ 8) in 2016. The relationship between pool 'fullness' and recent rainfall was strongest when using the mean rainfall of the survey month and previous two months. Therefore, this rainfall variable was included in the model, and estimates (Figure 7) represent a scenario with rainfall set to the mean value of this variable (~ 63 mm). The proportion of pools recorded as ‘largely full’ did not differ significantly ($P = 0.98$) between the 2005 field survey and estimates based on the 2001 aerial photograph.

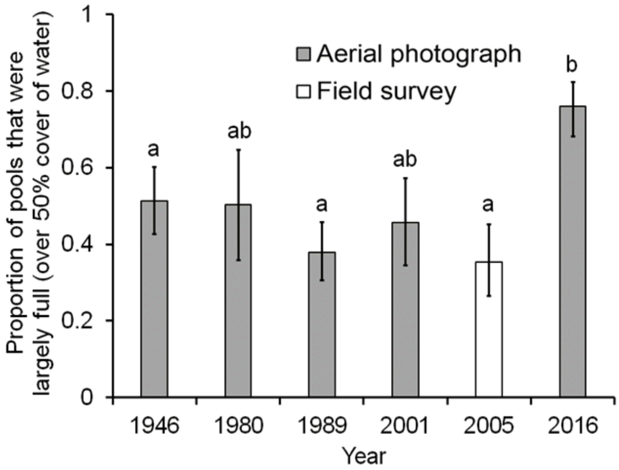

Figure 7. Changes in pool fullness over time. Estimated mean (and standard error) of proportions of pools per pool group that were largely full (over 50 % cover of water) in six survey years between 1946 and 2016. The Figure shows fitted year-wise means from the model reported in Table 3. The model includes recent rainfall as a covariate; year-wise mean estimates are for mean values of recent rainfall (~ 63 mm per month in the survey month and previous two months, averaged across Wick and Stornoway weather stations). Note that 2005 results were based on field survey, while results from other years were based on interpreting aerial photographs. Letters indicate year-wise means that were significantly different (Tukey-adjusted *P*-value < 0.05).

Table. 3. Statistical analysis of pool fullness. The *y*-variable was the number of pools in a group that were largely full, that is, had at least 50 % cover of water. The total number of pools in that pool group was the binomial denominator. Thus in effect, we analysed the proportion of pools that were largely full. The unit of analysis was pool group × year ($N = 5 \times 6 = 30$).

Effect type	Variable	Parameter estimate	Standard error	*P*-value	Covariance estimate (s.e.)
Fixed effects					
	Intercept	0.15	1.27	0.908	
	Recent rainfall [1]	0.016	0.016	0.343	
	Year	See Figure 7		<0.001	
Random effects					
	Pool group				0.40 (+/-0.31)

[1] Mean of monthly total rainfall in the month of the survey and the previous two months, averaged for two weather stations (Wick and Stornoway).

DISCUSSION

Key results and comparisons with other studies

The study achieved its aim of accurately quantifying bare peat extent, at least over half the Knockfin Heights plateau. Bare peat frequency (mean 2.7, 95 % c.l.s 2.2–3.2) was similar to values found in 2000 by Hancock *et al.* (2009) at four survey squares on Knockfin Heights (mean 2.2, s.e. 0.38), but around double their values from elsewhere in the Flow Country (0.92, s.e. 0.12, across 30 squares) (unpublished data). We estimated a study area mean bare peat cover of 10.5% (95 % c.l.s 6.8–15), with a mean longest length of bare peat per 100 m transect of 4.5 m (95 % c.l.s 3.0–6.2). We are not aware of comparable Flow Country figures; however, Hancock *et al.* (2009) found bare peat patches longer than 5 m in 69 % (s.e. 12) of (75) 500-m transect sections in four survey squares on Knockfin Heights, compared to only 9.0 % (s.e. 4.1) of (713) 500-m transect sections in 30 survey squares elsewhere in the Flow Country (unpublished data). Thus, large patches of bare peat were nearly eight times more common on Knockfin than elsewhere in the Flow Country. Similarly, in four 75–200 ha eroded peatland study sites in the Flow Country, Cummins *et al.* (2011) estimated bare peat extent at their Knockfin site as 3.1–5.9 times greater than at other sites. The more comprehensive (but also more qualitative) surveys reported by Matheson *et al.* (2003) suggest that, while the Knockfin plateau has some of the most extensive bare peat among Flow Country conservation sites (SSSIs), it is far from unique in this regard. The two SSSIs which cover the Knockfin plateau (Knockfin Heights, Rumsdale Peatlands) have, respectively, 91.5 % and 97.9 % of their area considered to hold 'significant' areas of bare peat (median 94.7 %) (Matheson *et al.* 2003). A further five of the 32 SSSIs reported on for this feature (of the 39 within the SAC) hold more than this median area of 'significant' bare peat. Our bare peat estimates were similar to those from an eroding study site in the Ladder Hills in southern Scotland (10.3 %: Cummins *et al.* 2011), but lower than estimates from highly-eroded northern England study sites at Rough Sike (17 %: Evans & Warburton 2010) and Bleaklow Plateau (34 %: Thom *et al.* 2016). An important caveat in using bare peat as a measure of erosion is that some deeply-incised areas may have relatively little area of exposed peat, but may have suffered considerable loss of peat. More detailed work would be needed if it was desired to accurately quantify rates of peat loss (see recommendations for future research below).

Our pool surveys confirmed that many pools were empty or nearly so, with over half of potential pool area not being covered by water in the 2005 field surveys. Although there was some evidence that the proportion of pools 'largely full' in 2016 was higher than in some earlier years, a quarter of pools were not full that year. Like Cummins *et al.* (2011), we found more extensive bare peat on flatter ground. Such areas are also where pool systems occur. Thus, we found a pattern that both pools and bare peat areas tended to occur in similar locations (flat ground), but recognise that longer-term work would be needed to better understand how these landforms are related. Some areas now largely comprising gullies and hags (e.g. Figure 2A) may have originated as eroding empty pools (e.g. Figure 2E), with subsequent erosion destroying the evidence (e.g. former pool shorelines) that might help clarify this. Alternatively, these features could have quite different origins. Although areas of bare peat are commonly treated as representing erosion, and are vulnerable (however they formed) to oxidation or erosion by wind and rain, the bare peat of exposed pool beds could in some cases reflect local peat accumulation (perhaps from eroding areas upstream), rather than erosion at that particular location.

We could not find quantitative published information concerning pool fullness in the Flow Country. One important study only considered permanent pools, i.e. containing water during the study period (Belyea & Lancaster 2002); another investigated pools marked on published maps (Lavers & Haines-Young 1996) without measuring how many held water.

Interpretation: possible causes of extensive bare peat and empty pools

A range of causes of peatland erosion, both natural and anthropogenic, have been suggested by past studies. At a particular site, erosion may reflect a variety of contributing factors, acting on different timescales (Tallis 1998). Here we consider potential causes in turn, and their plausibility at our study site.

Human influences potentially causing erosion include fire, grazing, atmospheric pollution and drainage (Mackay & Tallis 1996, Yeloff *et al.* 2006, Evans & Warburton 2010, Worrall *et al*. 2010). The last two seem unlikely here because Knockfin is remote from sources of industrial pollution and largely lacking in artificial drainage; this is similar to the more general view of drivers of peatland erosion across Scotland provided by Cummins *et al.* (2011). However, fire and grazing could have been important, whether in the last 100 years or much longer ago. There is unpublished palaeoecological evidence of past fires at Knockfin, perhaps increasing in recent centuries (S. Crowe, personal communication). Fire and grazing are often linked in the British uplands, where vegetation has long been traditionally burnt to improve grazing for sheep and deer. Lindsay (2010) considered fire the 'default trigger' of blanket mire erosion, unless evidence supports other processes at a particular site. Grazing and burning impacts in the Flow Country may have declined recently, due to (i) the influence of nature conservation since the 1990s (grant aid, surveillance, regulation, nature reserve establishment); and (ii) a change in sheep subsidies in 2005 leading to a 17 % decline in the number of breeding ewes in Less Favoured Areas of Scotland (mainly upland areas where agriculture is disadvantaged: https://ec.europa.eu/agriculture) during 2004–2010 (agricultural census data: www.scotland.gov.uk). Also, burning may have declined following widespread afforestation, because forestry plantations would be vulnerable to costly damage if an open-ground management fire escaped from control.

Other potential anthropogenic causes of erosion might act mainly at the edges of a bog. Peat cutting around the bog periphery can de-stabilise the peat mass and alter its hydrology, leading to erosion across a much larger area (Minayeva *et al*. 2017). Having found no signs of peat cutting on or near the plateau, we consider this mechanism unlikely at our study site. Alternatively, woodland clearance along watercourses can increase their erosive power, resulting in their cutting back into a plateau bog and triggering erosion (Tallis 1998, Ellis & Tallis 2001, Lindsay 2010); this process seems plausible here. Steeper slopes and stream-sides around Knockfin Heights probably held natural woodland in the past but almost all of this has been removed; extensive woodland clearance in Scotland commenced in Neolithic times (Smout 2002). The presence within recent centuries of wooded watercourses flowing from Knockfin Heights is implied by place names such as Allt Coire na Feàrna ('the stream of the alder [*Alnus glutinosa* (L.) Gaertn.] corrie'), and Allt Coille ('the woodland stream'), leading from the southern and north-eastern edges of the plateau, respectively. Both of these watercourses are almost treeless today, though the latter is shown as well wooded on the 1871 Ordnance Survey map.

Large herbivores (here, native Red Deer) can have trampling impacts on baulks separating pools; together with wind and wave action, this can break down the baulks, allowing higher pools to drain into lower ones (Bragg & Tallis 2001). Such baulk breakdown is one process where fire would seem an unlikely cause, at least within large pool systems. We observed some evidence of baulk breakdown processes in our study area. Whether this is seen as a natural or partly anthropogenic process depends on how natural we consider local deer numbers and movements to be. By the late 18th century in Scotland, deforestation and over-exploitation had dramatically reduced Red Deer populations, and their principal predators (Grey Wolves, *Canis lupus* Linnaeus, 1758) had been eliminated; however, Red Deer numbers recovered from the 19th century onwards (Harris & Yalden 2008). Widespread afforestation in the Flow Country since the 1980s has fenced deer out of large areas, changing their movements and potentially exacerbating trampling impacts in some areas (Lindsay *et al*. 1988). Cummins *et al*. (2011) considered that evidence of tracks created by deer trampling (visible in aerial photographs) was relatively pronounced at Knockfin, compared to other study sites. In recent decades, Red Deer management in the area (stalking, culling) has been increasingly influenced by nature conservation considerations, following designation of protected sites during the 1990s and 2000s (See METHODS: Study Area).

There are various potential natural causes of peatland erosion. Peat may creep down slopes at the edges of a peat-covered plateau (Evans & Warburton 2010), potentially de-stabilising the peat mass and triggering erosion. We did not see any clear evidence of this mechanism, although a similar process could have initiated erosion long ago, making it difficult to see the evidence today.

Some pools may be naturally unstable as long-term features. Belyea & Lancaster (2002) found that larger pools were deeper, and suggested that larger pools were older (following coalescence of adjacent pools over time). Thus, pools may deepen over time. If this deepening results from raising of the water table, then the pool bed may be better protected, e.g. from wave erosion. If, however, deepening takes place without any associated water table rise, then the depth of peat under the pool bed will decline. Ultimately, this could lead to a connection forming through the peat of the pool bed, allowing the pool to drain through to the underlying mineral soil. The vulnerability of pools on Knockfin to such a process is implied by Lindsay *et al*. (1988), who report that a pool on the plateau drained suddenly after a peat core was taken from its bed (see also Figure 2F). We found that larger pools were more often empty than smaller pools, which would be consistent with this process.

Among other natural processes linked to erosion, changes in climate could contribute to the triggering of erosion (Bragg & Tallis 2001, Evans & Warburton 2010), as follows. Higher winds or more regular frosts could increase erosive forces. Drier climates or shorter growing seasons could reduce the capacity of peatland vegetation to recover from impacts such as grazing, trampling and burning. Dry conditions over many years could lead to pools drying up and vegetating over. In our study area, the climate has perhaps recently become wetter, comparing mapped, modelled mean annual rainfall for 1961–1990 and 1981–2010 (www.metoffice.gov.uk). If rainfall has increased in a manner benefitting bog formation (e.g. more 'rain days': Lindsay *et al.* 1988), then this may have helped bog plants to re-colonise empty pool beds or grow across weak points in the baulks separating pools, perhaps helping explain the higher proportion of full pools in 2016 than in some earlier years. Set against this, however, is the fact that summers are getting warmer in the area (compare mean temperature for the summers of 1961–1990 and 1981–2010 at https://www.metoffice.gov.uk/public/weather/climate#?region=uk); such warming is seen as a key influence on peatland erosion (Gallego-Sala *et al.* 2010, Li *et al.* 2016). Also, extreme rainfall events occur more often as the climate warms (O'Gorman 2015), potentially increasing erosive losses of exposed peat (Evans *et al.* 2015). Finally, extreme wind storms have become more frequent in Britain in the last few decades (www.metoffice.gov.uk). Greater wave action might lead to the breakdown of baulks between pools and draining of upslope pools. Alternatively, wave action might tend to deepen pools. If the water table remains unchanged, such pool deepening could lead to a linkage developing through the peat of the pool bed into the underlying mineral substrate, causing the pool to drain (as discussed above).

Overall then, a range of factors, some natural, some human, operating at timescales of decades, centuries or millennia, may have contributed to the current state of the Knockfin Heights peatland. More detailed work, e.g. using palaeoecological approaches, would be needed to better understand the course of erosion over time at the site, and hence better point to plausible causes. However, given the apparently recent timing of some pool recovery, we tentatively suggest that this may have been influenced by some of the more recent changes in the area such as sheep reductions, recent climatic changes, and/or changes in deer and prescribed fire management following conservation designation.

Study limitations and recommended future research

One study limitation is the lack of a repeat of the 2005 ground survey, which would allow accurate measurement of the rate of change of bare peat extent. Cummins *et al.* (2011) inferred from aerial images that spontaneous recovery may be taking place at Knockfin Heights, and recommended follow-up field surveys. Our baseline ground survey offers the potential to measure changes in the locations of bare peat edges to within a few cm. Our field surveys characterised fine-scale vegetation using a frequency method likely to be highly repeatable and robust to changes of observer (see METHODS). For every vegetation frequency sample point, we took a digital photograph in 2005; these points could be accurately relocated. Repeating this work would allow measurement of fine-scale changes after 12 or more years, in relation to a range of baseline conditions. This could help identify thresholds which, when surpassed in the baseline state, lead to a different trajectory. Such thresholds, and linked processes such as encroachment of peatland vegetation onto bare ground, have been identified as key peatland research priorities (Belyea & Baird 2006, Evans & Warburton 2010), in tune with wider thinking on the importance of threshold processes to ecosystem management (Suding & Hobbs 2009). We consider that a repeat ground survey would be the highest-priority (and perhaps simplest) piece of future research work to carry out at the site.

In addition to follow-up ground surveys, other approaches - experimental (Holden & Burt 2002) or palaeoecological (Crowe *et al.* 2008) - may allow a better understanding of causes of erosion and recovery. Experimental deer exclosures would allow better measurement of deer effects on the plateau (as used on the Monadhliath Hills: Cummins *et al*. 2011), although such exclosures could themselves impact the fragile habitats of the plateau by concentrating trampling around their edges. Palaeoecological studies could illuminate the long-term history of the site, helping to explain its current state. The vegetation history of the area could be reconstructed from peat cores using pollen and plant macrofossils together with other indicators like testate amoebae (Creevy *et al*. 2018) and charcoal fragments (Mackay & Tallis 1996). Cores taken from terrestrial deposits (Ellis & Tallis 2001) could, therefore, allow trends in climate, management, and erosive processes to be inferred. Cores from lake beds (e.g. Rhodes & Stevenson 1997) offer less potential in the area, due to the lack of deep lakes receiving water from the plateau, although local lake-bed sediments could provide useful information on the history of erosion in the wider area. Maps could also offer a useful historical perspective. British and Irish peatlands are unusual among the world's peatlands in having been accurately mapped in the 19th century at a scale of 1:10,560 (Ordnance Survey First Series: e.g. National Library of Scotland, www.nls.uk). In the Flow Country, these maps date from 1868–1873; as well as mapping pool systems and streamside woodland in detail, they record the heights of some pool banks and water surfaces (on specific dates) to an accuracy of ~ 3 cm (stated as 0.1 feet).

Of these various options, for understanding causation, we consider that reconstructing past water tables (e.g. using testate amoebae) would be one of the most promising future research avenues to develop. For understanding how best to manage the site, we consider that deer exclosures would be particularly informative in illuminating the effects of deer, which are arguably the most important factor under some degree of management control in intact Flow Country peatlands.

For this study we had to restrict the range of plant species surveyed, due to the short time available. However, vegetation work for future ground surveys should include a wider range of taxa. Prime among these would be *Eriophorum* species, which are increasingly viewed as keystones during the spontaneous re-vegetation of damaged peatlands (Evans & Warburton 2010). In the Flow Country, *E. vaginatum* was one of the most strongly responding species during restoration of a site damaged by afforestation (Hancock *et al.* 2018). *Eriophorum* species can facilitate the establishment of *Sphagnum* species from spores (Sundberg & Rydin 2002). This could be an important process at sites like Knockfin, where *Sphagnum* has a low frequency of occurrence. It would also be useful to measure deer dung deposition rates to obtain an index of deer activity at a fine temporal and spatial scale (Cummins *et al.* 2011). In future work it would be valuable to make a more detailed appraisal of slope along the monitoring transects, in relation to the wider landscape slope, given that this might significantly affect both processes (such as water flows) and outcomes (such as the rate of expansion of vegetation cover over bare peat areas). It would be valuable to collect local rainfall data on the plateau itself, alongside ground-based measurements of pool depth, pool area and changes in water level, to be able to more accurately account for the effect of recent rainfall on pool fullness rather than (as here) relying on relatively distant weather stations. Finally, future ground surveys should consider categorising and mapping landforms according to the typologies of Bower (1961; see also Evans & Warburton 2010, pages 76–80), Lindsay (1995) or Evans & Warburton (2010, pages 164–167).

While most landowners cooperated with our study, survey work was restricted by others who refused access to around half of the plateau, particularly over the eastern part of the area. This could have introduced a bias into our results, in that the more sheltered lee side of the plateau was not surveyed. Such access refusals have been found elsewhere to bias the range of sites studied, as found in the UK uplands by Sim *et al.* (2005). The recent (2003) introduction in Scotland of a legal right of responsible access means that simple walk-over surveys compliant with the access code (www.outdooraccess-scotland.com) would now be legitimate over the whole plateau, if formal permission could not be obtained. We were also unable to complete all survey squares in the southwestern part of the plateau, due to its remoteness. Therefore, our work focused on western and northwestern areas. Future work should aim to cover the whole plateau. Given the potential importance of wind and frost action in affecting erosion (Bragg & Tallis 2001), aspect could affect the relative importance of different erosive processes.

Stratification of sampling work into 'more eroded' and 'less eroded' strata was intended to ensure that a wide range of degrees of erosion or bare peat were included in the study. Although their vegetation differences appeared quite minor, the two strata differed strongly (roughly by a factor of two) in the extent of bare peat (see RESULTS). The wide range of bare peat values among the survey squares (e.g. Figure 5) may have helped elucidate the correlates of bare peat cover. However, the more extreme bare peat values were still quite rare in the dataset. Given that such areas are arguably of greatest interest, future work could consider employing a relatively higher sampling rate in the 'more eroded' stratum.

As well as addressing questions around habitat condition for nature conservation - the context of the current study - future work should consider the wider climate and water quality implications (Thom *et al*. 2016) of habitat trajectories at eroding peatlands like Knockfin. A powerful approach is to complement ground-based surveys with remote sensing (Lindenmayer *et al.* 2018). Beyond the conventional aerial photography exploited here, there are new opportunities for detailed mapping (including beyond the visual spectrum) using drones (Lovitt *et al*. 2017). Remote methods offering particular potential include

LiDAR (Cummins *et al.* 2011), satellite imagery (Lees *et al.* 2018) and InSAR, a recently developed approach which is currently being used at sites in the Flow Country (e.g. Alshammari *et al.* 2018).

Management implications

Generally, reductions in grazing and burning are predicted to reduce erosion (e.g. Li *et al.* 2016). However, these processes are not currently at high levels at Knockfin Heights, partly owing to nature conservation influences. We recorded long-term mean Red Deer densities around 4.6 animals km^{-2} at the northern end of Knockfin Heights, 31 % of a commonly recommended maximum of 15 km^{-2} (Cummins *et al.* 2011). At body weights around 70–100 kg (Harris & Yalden 2008), our densities equate to around 300–450 kg km^{-2}. Li *et al.* (2016) considered 50 sheep km^{-2} to be 'light' grazing on UK peatlands; at sheep body weights around 40–60 kg (e.g. Annett *et al.* 2011) this equates to around 2000–3000 kg km^{-2}. Thus, our estimate of ungulate biomass density was ~10–20 % of that considered to be 'light grazing' elsewhere on UK peatlands. In the wider Flow Country, Headley (2006) considered ungulate grazing and trampling pressure unlikely to damage nature conservation status, except perhaps in a few localised areas. In any case, some pattern of Red Deer usage, effect and, indeed, impact is natural within peatland ecosystems in this region, this being a native wild animal.

Certain 'no regrets' management options could be considered; that is, options which might provide benefits anyway, whether or not they are strictly supported by the evidence in relation to this particular issue. Deer movements have been altered by extensive deer fences, mainly built in the 1980s to protect forestry plantations. Such fences can concentrate deer routes, leading to trampling impacts (Lindsay *et al*. 1988). Some forestry plantations undergoing restoration as blanket bog (e.g. Creevy *et al*. 2018, Hancock *et al*. 2018) retain redundant fences, which could be removed. Another possible 'no regrets' option relates to riverine and streamside native woodland, which still persists in some parts of the Flow Country. Promoting recovery of this woodland type could reduce the erosive power of these watercourses, with potential benefits to summit blanket bogs.

Ultimately, clear management recommendations remain elusive, while the origin - natural and/or anthropogenic - of the erosion features on Knockfin Heights remains uncertain. If natural in origin, these features are worthy of conservation (Lindsay *et al.* 1988), especially if they do not generate significant soil carbon losses. We welcome the increasing impetus and funding in Scotland supporting active restoration of damaged peatlands. However, where strong spontaneous recovery is taking place, active restoration management may be a poor use of such funds (Minayeva *et al.* 2017), particularly at remote sites like Knockfin, where management would be relatively expensive. Therefore, decisions over whether to implement restoration management should follow a careful assessment of the likely origins and trajectories of erosion, ideally exploiting repeatable quantitative baselines such as that provided by the current study.

CONCLUSIONS

Our study has quantified the extent of bare peat and exposed pool beds on the Knockfin Heights plateau in the Flow Country. At a mean of 10.5 % cover, bare peat was found to be as extensive at Knockfin as it is at highly-eroded peatlands in southern Scotland and northern England. Among the extensive pool systems at Knockfin, over half the investigated pool area was found to be dry during ground surveys in 2005. However, air photographs suggested that (after accounting, approximately, for rainfall variation), more recent years have seen some increase in the mean proportion of pool bed area that is covered in standing water. Bare peat, whether from exposed pool beds or other features, was concentrated on the flattest ground, with areas of zero slope estimated to have around ten times more bare peat cover than ground at ~ 4 ° slope, potentially suggesting a link, on flatter areas, between the dynamics of pool systems and wider areas of bare peat.

Management decisions at the site depend on both the origin and the trajectory of the bare peat features. Erosion of anthropogenic origin showing no spontaneous recovery would merit restoration management. Conversely, if bare peat areas were shown to be natural in origin or recovering spontaneously, this would support non-intervention. We recommend the following priority actions to inform these questions: (i) a repeat of the 2005 ground survey, with some modifications and improvements as suggested here; (ii) an investigation of the history of the site using palaeoecological approaches (e.g. testate amoebae, charcoal fragments) to reconstruct water table and fire history; and (iii) the construction of deer exclosures, with subsequent vegetation monitoring, to determine whether deer at current (low) densities are influencing rates of re-vegetation of bare peat.

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Appendix: Survey squares and transects, Knockfin Heights field survey, 2005. 1 m grid references (British Ordnance Survey National Grid) - all start with 100 km grid letters "NC". The full electronic data are held as Excel spreadsheets in a folder entitled 'Knockfin data for *Mires and Peat* submission, 2018', on RSPB servers at Forsinard, Inverness and Edinburgh, and can be obtained by request from the authors or their successors.

Square code	Stratum (L='less eroded', M='more eroded')	Square south-west corner	100 m transect start point	Transect magnetic bearing (degrees)*	Actually surveyed?	Estimated slope (degrees)
S2	L	90000 33000	90200 33400	137	No	2.8
S4	L	90500 32500	90700 32700	162	No	1.6
S5	L	90500 31500	90900 31800	317	No	2.8
S8	L	91000 32500	91200 32900	151	Yes	1.2
S10	L	91500 36500	91900 36900	336	Yes	3.6
S12	L	91500 34500	91800 34800	297	Yes	1.2
S14	L	92000 36500	92200 36600	359	Yes	2.8
S19	L	92500 36500	92600 36800	238	Yes	2.8
S20	L	92500 31000	92800 31400	114	Yes	3.6
S21	L	92500 29000	92700 29300	38	No	2.8
S26	L	94000 37000	94400 37100	156	No	0.8
S27	L	94000 36500	94300 36700	199	Yes	1.6
S28	L	94000 29000	94300 29300	136	No	3.6
S29	L	94500 37000	94900 37400	207	Yes	1.2
S33	L	96500 36500	96800 36800	84	Yes	1.6
E4	M	91000 33000	91300 33300	349	Yes	0.4
E5	M	91000 32000	91400 32400	183	Yes	2.4
E6	M	91500 36000	91700 36200	283	Yes	2.0
E11	M	92000 35500	92200 35800	360	Yes	2.8
E13	M	92000 34500	92100 34900	138	Yes	0.8
E14	M	92500 36000	92700 36100	51	Yes	3.6
E17	M	92500 30000	92600 30200	35	Yes	1.2
E19	M	93500 38000	93900 38400	169	Yes	2.0
E21	M	94000 37500	94300 37900	158	Yes	2.0
E22	M	94500 38000	94900 38300	247	Yes	0.0
E23	M	94500 37500	94800 37800	302	Yes	1.2
E24	M	94500 36500	94700 36700	195	Yes	1.6
E27	M	95000 36500	95100 36600	5	Yes	0.8
E28	M	95500 36500	95900 36600	204	Yes	2.0
E30	M	97000 36500	97200 36800	258	Yes	1.6

* At the time of the survey, magnetic north was about 4° 41' west of true north.