A regime shift from erosion to carbon accumulation in a temperate northern peatland

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Abstract

1. Peatlands are globally important ecosystems but many are degraded and some are eroding. However, some degraded peatlands are undergoing apparently spontaneous recovery, with switches from erosion to renewed carbon accumulation – a type of ecological regime shift.
2. We used a palaeoecological approach to investigate and help understand such a switch in a blanket peatland in North Wales, UK.
3. Our data show: (i) a rapid accumulation of new peat after the switch from the eroding state, with between 5.2 and 10.6 kg m^{-2} carbon accumulating since the beginning of the recovery which occurred between the late 1800s and early to mid 1900s CE, with an average carbon accumulation rate in the new peat between 46 and 121 g C m^{-2} yr^{-1}; (ii) three main successional pathways in peat-forming vegetation; and (iii) hydrological changes with an increase to moderately high water tables after the switch that promoted new carbon accumulation as well as protecting vulnerable old carbon. External factors, including changes in climate and industrial activity, can only partially explain our results. Following previous studies, we suggest that internal ecosystem processes offer a substantial part of the explanation and interpret the switch to renewed carbon accumulation as a bifurcation-type tipping point involving changes in the physical form of the eroded landscape.
4. *Synthesis:* Our long-term ecological data reveal a switch from a degraded peatland with active erosion and loss of carbon to a re-vegetated, wetter peatland accumulating carbon. The switch can be interpreted as a bifurcation tipping point. We suggest that external factors such as climate and pollution levels are important for setting suitable boundary conditions for peatland recovery, but internal mechanisms can explain the change in peatland state. Our study is the first of its kind to apply tipping-point theory to the internal mechanisms linked to peat erosion and recovery and may help improve understanding of the trajectories of other peatlands in a changing climate.

Keywords: peatland erosion, carbon accumulation, blanket bog, regime shift, bifurcation tipping point, spontaneous recovery, peatland dynamics, palaeoecology and land-use history

Introduction

Peatlands are globally important ecosystems (Page and Baird, 2016). Because they are large carbon (C) reservoirs – storing twice as much C as all the world's forests – there is growing interest in how peatlands function, so that their behaviour and C-sink strength under a changing climate and different land uses may be modelled (Page and Baird, 2016). It is widely documented that many peatlands are degraded. In blanket peatlands degradation frequently occurs as erosion, which results in loss of C and loss of important peatland habitats over centennial timescales (Gallego-Sala and Prentice, 2013). It is sometimes assumed that, once initiated, erosion will continue until all of the original peat mass has been removed (e.g. Wishart and Warburton, 2001). However, some blanket peatlands are undergoing apparently spontaneous recovery, switching from erosion to renewed C accumulation. Recent work has shed light on factors affecting peatland initiation (e.g., Morris et al., 2018), their responses to climate change (e.g., Charman et al., 2013), and the relative importance of autogenic and allogenic factors in controlling changes in the composition of peatland vegetation and rates of peat accumulation (e.g., Belyea and Baird, 2006; Belyea, 2009). Peatland initiation and re-initiation (the latter being synonymous with re-vegetation as shown in Fig. 1) are types of ecological regime or state shift (Beisner et al., 2003; Folke et al., 2004; Andersen et al., 2009), as is the switch from an intact peatland to an eroding one. In ecology, a regime shift may be defined as "a sudden shift in ecosystem status caused by passing a threshold where core ecosystem functions, structures and processes are fundamentally changed" (Andersen et al., 2009). Compared to many ecosystems, such as shallow lakes and coral reefs where the causes of sudden changes have been revealed (see Folke et al., 2004), regime shifts in peatlands are poorly understood.

Blanket peatlands occur in oceanic and hyper-oceanic high-latitude regions where precipitation (mostly rainfall) is abundant throughout the year, causing water-logged conditions in which peat-forming plants (principally *Sphagnum* mosses and the cotton sedges (*Eriophorum* spp.)) thrive (Lindsay, 2010; Gallego-Sala and Prentice, 2013). Although some blanket peatlands may have formed in response to forest clearance in the Neolithic and Bronze Age, or earlier (e.g., Tallis, 1998), there is strong evidence they are also natural phenomena as a result of climate shifts or soil-forming processes (Lawson et al., 2007; Tipping, 2008; Gallego-Sala et al., 2016), with initiation dates extending as far back as 8000–12000 cal BP (calibrated years before present, with present defined as 1950). Erosion has been a feature of blanket peatlands in the UK for centuries and was reported as long ago as the early 19th Century (e.g., Aiton, 1811). Erosion may be associated with the developmental phase of the peatland – i.e., it may be an autogenic phenomenon – but may also arise from natural and human-induced changes in environmental conditions (climate, land use, and atmospheric pollution) (Conway, 1954; Bower, 1962; Bowler and Bradshaw, 1985; Bragg and Tallis, 2001). Blanket peatland erosion is characterised by the formation of gullies (Bower, 1962; Wishart and Warburton, 2001; Clement, 2005), and gully networks may be linear (gullies sub-parallel to each other), dendritic, or anastomosing. The last type of landscape is often termed a 'hagged peatland' (Figs 1 and 2), and usually occurs on lower-gradient areas such as hilltops and plateaus ('hag' or 'hagg' usually denoting the 'islands' of un-eroded peat found between gullies; e.g., Bowler and Bradshaw, 1985; Foulds and Warburton, 2007).

Many erosion complexes are showing signs of recovery, with peat-forming vegetation re-establishing, (i) on previously-eroding bare-peat surfaces, (ii) in areas where eroded peat has been deposited, and (iii) in areas of mineral ground exposed by the complete removal of peat in gully bottoms (Fig. 1).

However, it is unclear whether re-vegetation is a temporary phenomenon in a landscape where erosion will ultimately remove all of the peat mass (Wishart and Warburton, 2001) or whether it can lead to the re-stabilisation of the peatland and the infilling of gullies (Evans and Warburton, 2007; Crowe et al., 2008). Recent work by Harris and Baird (2018) has shown that the location of the principal plant types in a recovering hagged peatland in North Wales (UK) can be predicted with some accuracy using hydrological and micro-meteorological metrics derived from a fine-scale topographic model of the peatland. Although useful in indicating the conditions required for re-vegetation to occur, such an approach is necessarily 'static' because it does not reveal when re-vegetation started or those factors unrelated to the topography that may have been involved in the change in system state: the switch from erosion to renewed C accumulation. Similarly, successional changes following the switch, and the degree to which re-vegetation has led to C accumulation, cannot be obtained from a study of current vegetation distribution. To address these limitations, and in the absence of long term monitoring, we took peat cores from the same peatland studied by Harris and Baird (2018) and used a high-resolution palaeoecological approach similar to that of Taylor et al. (2019) and Swindles et al. (2015) to reconstruct the recent ecohydrological history of the peatland. Specifically, we sought to address the following key questions: (i) When did the recovery start and what are the possible mechanisms?, (ii) What successional processes have occurred?, (iii) Have water levels relative to the ground surface changed or remained stable since the start of the recovery?, and (iv) How much peat (and associated C) has accumulated since recovery started?

Figure 1: Landscape of erosion and re-vegetation. Photos of the study site (see Fig. 2) showing a) Actively-eroding haggs and inter-hagg gullies. The haggs are ~1 m high. b) Re-vegetation in wider gullies between haggs, acting to stabilise previously-eroding peat. c) Erosion of gully down to the mineral substrate (see mid and lower right of image) with re-vegetation, here a combination of mostly *Juncus* spp. and *Sphagnum* spp.

Materials and methods

The study was carried out on a hagged area of blanket bog in the Migneint in the Upper Conwy catchment, North Wales, UK (latitude 52.97°N, longitude 3.84°W). The peatland site (Fig. 2) lies at an altitude of approximately 500 m above sea level. Average rainfall is 2100 mm yr^{-1}, and average January and July temperatures are 2.2 and 12.8 °C respectively (Green et al., 2017; Swindles et al., 2016). The peat overlies thin glacial deposits, and Cambrian mudstones and Ordovician siltstones (Lynas, 1973). Little is known about the historical management of the area other than the site has been used for grazing, and some areas just beyond the study site were drained with shallow (~50–70 cm deep) ditches constructed between the 1940s and 1970s (Swindles et al., 2016).

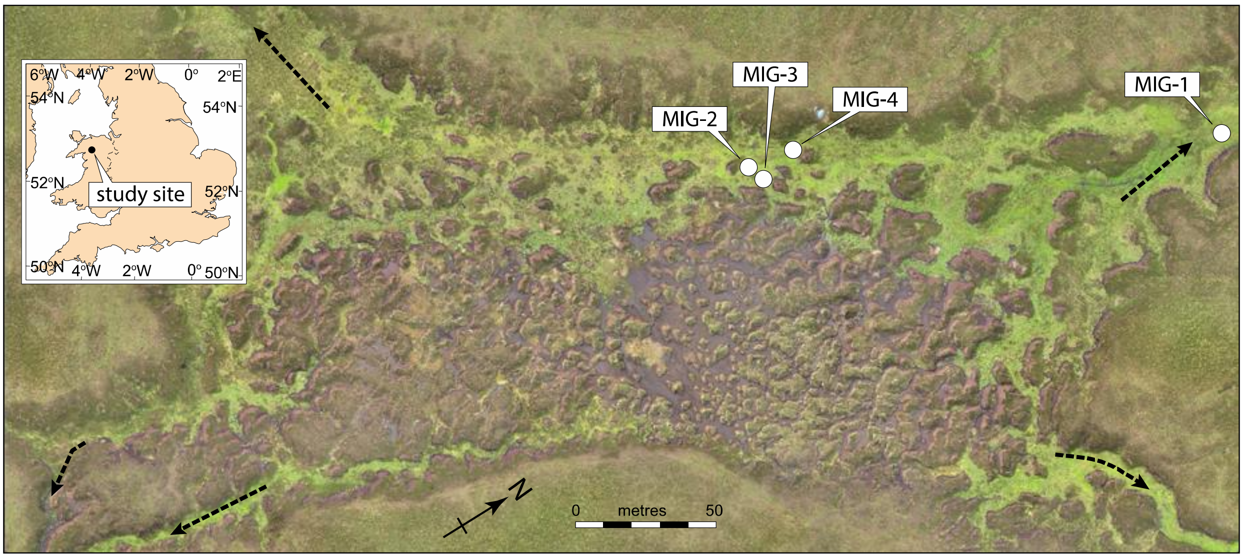


Figure 2: The study site. The aerial image shows actively-eroding haggs and inter-hagg areas that have re-vegetated. The locations of the peat cores (MIG1, MIG2, MIG3, MIG4) and the main drainage routes (dashed arrows) are also shown. The inset map shows the location of the study site in the British Isles.

Erosion on the gently-sloping hilltop has resulted in a network of erosion gullies interspersed with peat haggs. The haggs are typically topped with *Calluna vulgaris* (L.) Hull. and slope in a SW-NE direction, similar to the prevailing wind direction. Bare, actively eroding peat remains in some places between the haggs, with the mineral substrate sometimes visible, while in other areas the vegetation has regenerated mostly with *Juncus effusus* L., *Sphagnum* (particularly *S. fallax* H. Klinggr. and *S. cuspidatum* Ehrh. Ex Hoffm.), cotton sedges (*Eriophorum vaginatum* L. and *E. angustifolium* Honck) and *Polytrichum commune* Hedw. There are two main re-vegetation areas (Figs 1 and 2): wide gullies in which new peat has established mostly on mineral ground, and the areas around the edge of the main hagged area, where new peat has established in some places on bare, formerly-eroding peat, and in some places on mineral ground. We collected four peat cores from the two areas (MIG1 from a wide gully and MIG2, MIG3 and MIG4 from the edge of the main hagged area, Fig. 2). Large diameter cores of 15 cm were collected using the scissor method described by Green and Baird (2013), to avoid compaction and disturbance of the loosely consolidated peat. The MIG1 and MIG4 cores were collected from areas where visual evidence suggested all of the old peat had been stripped away and new peat had formed directly on mineral ground: exposed peat faces at the edges of the gully near to the coring locations showed sub-peat mineral material (Fig. 1c), while depth probing with an auger suggested the peat was 40–50 cm deep. However, the 40–50 cm cores did not reach the mineral material and neither could we be certain that there were no traces of old peat between the new peat and mineral material, although we think it is likely that new peat formed directly on mineral material at MIG1 and MIG4. More information is provided on this point in the SI. Initial visual inspection of MIG2 and MIG3 cores suggested that they consisted of newly accumulated peat on top of a former erosion surface, which marked the top of old, Holocene peat. The dating (discussed below) was used to check this interpretation. The dating was also used to ascertain whether there was old peat at the base of cores MIG1 and MIG4 or whether the samples contained new peat only.

We analysed the cores in several ways to address the research questions. To help answer the first question, we dated the cores using ^{210}Pb (contiguous 1 cm intervals) and AMS ^{14}C. ^{210}Pb ages were calculated using the Constant Rate of Supply model (Appleby and Oldfield, 1978; Appleby, 2001). We obtained additional dates using AMS ^{14}C, particularly to confirm the date of un-eroded peat below the newly-forming peat. We applied the age-depth model package rbacon v.2.4.2 using Bayesian modelling to integrate ^{210}Pb and ^{14}C dates (Blaauw and Christen, 2011, R Core Team 2020). Contiguous, 1-cm down-core samples were analysed for plant macrofossils to characterise vegetation development and address the second question. The method used follows Gałka et al. (2017), with seeds and fruits assessed using counts, and other plant remains (e.g., leaves, roots) estimated as volume percentages. To answer the third question, we analysed 1-cm thick samples taken every 4 cm for testate amoebae to reconstruct changes in water-table depth. Analysis followed Booth et al. (2010), and the water table reconstruction was calculated using the pan-European transfer function (Amesbury et al., 2016). To answer the fourth question, we measured the dry bulk density and the C content of the peat for contiguous 1-cm samples. Analysis for peat dry bulk density, organic matter content through loss-on-ignition, and the carbon-nitrogen quotient through elemental analysis followed Chambers et al. (2011). Apparent C accumulation rates in the newly-formed peat were calculated by multiplying the mass per unit area of each depth increment by its C concentration and then dividing by the difference between the bounding dates of the increment, as obtained from the age-depth model (Tolonen and Turunen, 1996). Apparent C accumulation rates were not calculated beyond the limit of ^{210}Pb dating. Full information on the methods is provided in the SI.

Results

Below, we discuss the results in relation to each of the research questions and with reference to Fig. 3.

When did spontaneous recovery start and what are the possible mechanisms?

Radiocarbon dates for MIG2 and MIG3 confirm that the lower part of the peat profile in each case formed around 7000 cal BP, and represents the remains of the original Holocene peat mass, with younger peat from that mass removed by erosion. The age-depth model indicates that the oldest ‘new’ peat formed in 1875 CE ± 45 years in MIG2 and 1906 CE ± 23 years in MIG3 (both record the boundary between old and new peat). Taking a conservative approach, and assuming the basal new peat may represent ‘false starts’ in the system before continued re-vegetation (see conceptual model and accompanying discussion below), spontaneous recovery started sometime between 1875 ± 45 and 1931 ± 21 for MIG2, and between 1906 ± 23 and 1934 ± 10 for MIG3. The oldest peat in MIG1 is 1922 ± 14 years, and in MIG4 it is 1950 CE ± 26 years, suggesting all of the peat at these locations is new and that re-vegetation started at, or sometime before, these dates. However, because the boundary between the new peat and underlying mineral material or old peat was not recorded at MIG1 and MIG4, estimates of the timing of the switch are conservative. Combined, the data suggest that the majority of the new peat started accumulating from around the late 1800s and early to mid 1900s CE, with an apparent acceleration in the last 40 years (see below for discussion on C accumulation). We searched for regional events occurring around the date of spontaneous recovery and this onset of more rapid C accumulation, and briefly outline them below.

Change in climate: Climate warming occurred in the mid- to high-latitudes in the Northern Hemisphere after the Little Ice Age from the middle of the 19th century to the present, which reversed the long-term cooling trend of the past 5000 years (Masson-Delmotte et al., 2013; Ljungqvist et al., 2019). The Central England Temperature record – the longest record in the UK – shows an increase in annual average temperature of approximately 1.5 °C since 1850 CE (Parker et al., 1992), and centennial trends show increased temperatures in all seasons since the record began in 1659 CE, with greatest increases in winter and autumn, and associated increases in growing degree days (Kendon et al., 2019) (see SI). The timing of the post-Little Ice Age and 20th century warming broadly coincides with the switch seen in our data, suggesting that there may be some climatic influence on the recovery and increase in rates of C accumulation.

Change in industrial activity: There is a long history of below-ground and open-cast mining in north-west Wales, with mining of the Ordovician slate veins pre-dating the Medieval Period. Activity increased substantially at the end of the 18th century; quarry output peaked in the late 19th century, decreased by the 1920s, and the majority of quarries had closed by the 1960s (Prichard, 1942; Hughes et al., 2016). The nearest quarry to the site (~ 1.5 km away) was worked from 1870 for about 50 years. Given the timing of the regional industrial activity, it is possible that dust fertilisation from mining contributed to the shift to peat accumulation. Elemental analysis of the peat cores (not shown here) revealed low concentrations of Fe and K in the old peat and an increase in the new peat, possibly coinciding with quarry activity.

Change in grazing regime: Sheep farming in the Welsh uplands was a major industry from the 13th century onwards. Since the 1850s there has been an increase in Welsh sheep populations from 2.5 million in 1867 to a peak of ~12 million at the end of the 1990s (Welsh Government, 2018). Estimation of wild animal densities are not available, and although cattle are important domestic livestock in Wales, they are less commonly grazed in the uplands. Given that sheep densities were still increasing in the 18th, 19th and 20th centuries, changes in grazing are unlikely to have contributed to the shift to peat accumulation and we dismiss it here as a possible primary factor in the switch.

Other possible external factors were also dismissed. For example, there is no evidence of historical peat cutting in the region until the 1970s (Evans et al., 2008), and there is low charcoal abundance in the macrofossil data and no unconformity in peat accumulation (beyond the unconformity observed between old and new peat), indicating minimal impact of fire on peat accumulation. Therefore, apart from potential climate amelioration and a fertilisation effect from quarry activity, we could not identify any obvious cause for the switch. Further details on external factors are provided in the SI.

What successional processes occurred?

We identified three successional pathways in the macrofossil data that differed according to whether the re-vegetation occurred on mineral ground or bare, formerly-eroding peat. MIG2 and MIG3 record the same broad successional pathway after the shift, with *E. vaginatum* succeeded by *S. fallax* with a later increase in Cyperaceae (Fig. 3). Both cores represent re-vegetation on formerly-eroding bare peat. MIG1 and MIG4 represent re-vegetation on mineral ground (probably – see above) and both record broadly similar successional pathways, but with a difference in dominance. MIG1 shows a transition from *Juncus* with *S. fallax* to dominance by *P. commune*; and MIG4 records a transition from herbaceous species to dominance by *S. fallax*, with an almost continuous presence of *Juncus*. The complete macrofossil record for each core is provided in the SI.

Have water levels changed or remained stable since spontaneous recovery?

When re-vegetation first occurred, water tables were relatively deep at ~25–10 cm meaning they were deep within the existing peat mass or fell into the mineral material below. After recovery, water tables were moderately high and stable with some evidence of standing water (e.g. *Arcella discoides* (Ehrenberg 1843) in MIG4) and nutrient-poor conditions, typical of blanket peatlands. Reconstructed water-table depths at the present surface are between 7 and 15 cm. There are very few indicators of minerotrophic conditions within the cores, although the presence of *Centropyxis cassis* (Wallich 1864) towards the base of MIG1 may indicate greater mineral influence, coinciding with the herbaceous/*Juncus*-dominated interval in the macrofossils and possibly reflecting new peat formed on exposed mineral ground. There is poor preservation, low diversity and an absence of testate amoebae within the old peat or at the boundary between old and new peat in MIG2 and MIG3, perhaps suggesting oxidation of the upper part of the old peat immediately prior to the switch.

How much peat and associated C has accumulated since spontaneous recovery?

There was a rapid accumulation of peat after the switch, with a weighted average C accumulation rate for all cores between 46 and 121 g C m^{-2} yr^{-1} from the earliest date of the switch (MIG1 = 109.7, MIG2 = 61.8, MIG3 = 46.1, MIG4 = 121.0 g C m^{-2} yr^{-1}). The amount of new C in our cores since the switch (C addition) ranges from 5.2 kg m^{-2} (MIG3) to 10.6 kg m^{-2} (MIG1), with MIG2 and MIG4 containing 8.8 and 8.4 kg m^{-2} respectively. Where newly-formed peat is underlain by older remnant peat, as is the case at MIG2 and MIG3, the decay losses from this older peat (even if protected to some extent by a rising water table following the switch) need to be combined with the rates of new peat addition to obtain *net* rates of C accumulation. However, such losses are difficult to estimate retrospectively (Young et al., 2019). At MIG1 and MIG4, it is likely that all old peat was eroded away before the new peat developed (see above and SI). Therefore, our figures for net C addition for these locations are the same as new C accumulation, and are conservative because we did not reach the base of the new peat. Uncertainty in the age models for all four cores means that there is associated uncertainty in the C accumulation rates (see SI). Nonetheless, combined, our data show that new peat is rapidly accumulating, while at the same time, older remnant peat is now protected from rapid aerobic decay by a higher water table.

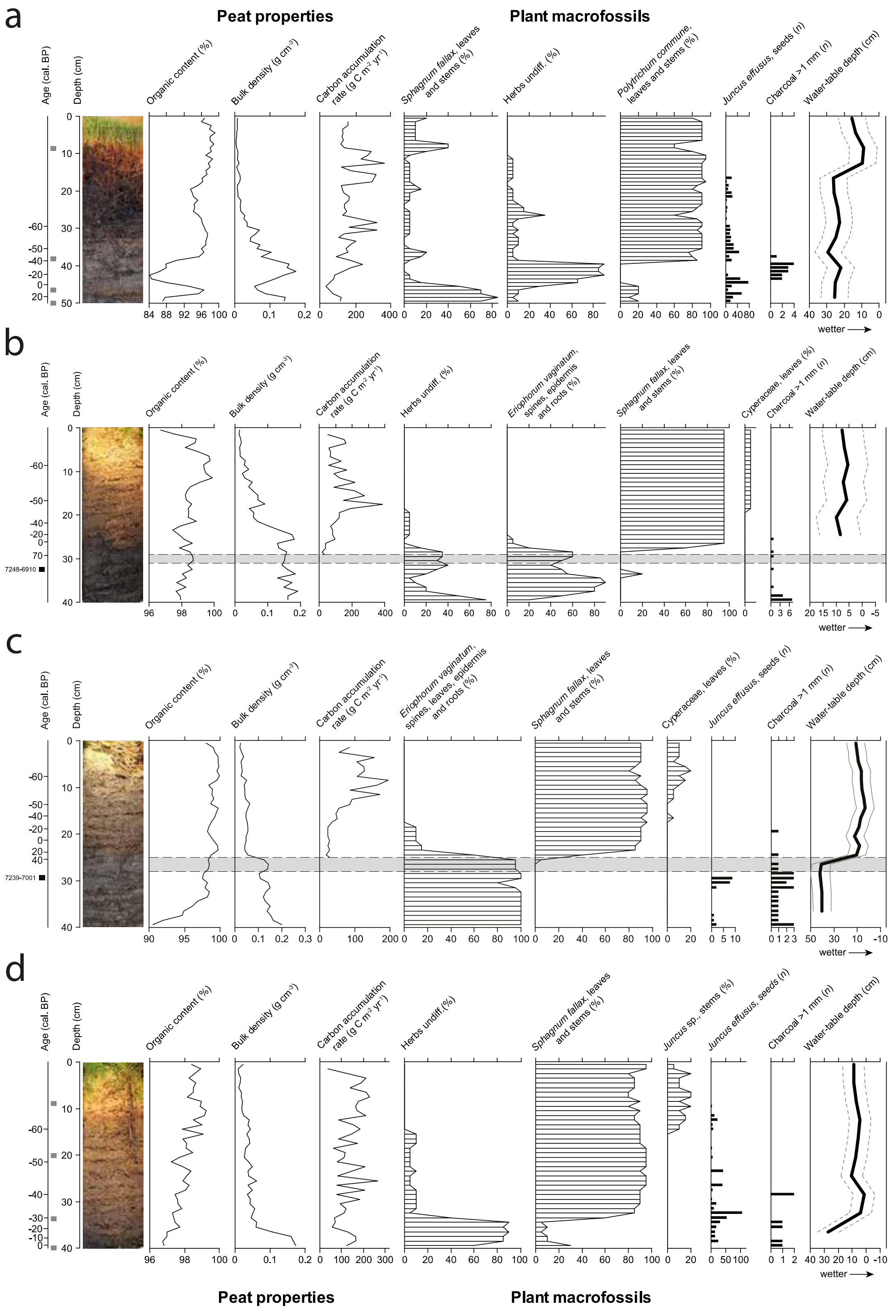


Figure 3. Peat characteristics and palaeoecological data for a) MIG1, b) MIG2, c) MIG3, d) MIG4. For each core, organic matter content (%), dry bulk density (g cm^{-3}) and C accumulation rate are shown alongside the core photo; also shown are key taxa from the plant macrofossil data (displayed as percentage or count data); charcoal abundance; and water-table depth reconstructions from testate amoebae. All data are plotted against depth (cm) with an age scale (cal. BP) shown for reference. The dashed grey bar in MIG2 and MIG3 represents the boundary between old (below) and new (above) peat. The location of the radiocarbon dates are shown by squares (black for old, Holocene peat in MIG2 and MIG3; and grey for recent peat in MIG1 and MIG4).

Discussion

Palaeoecological studies from peatlands are common but these typically focus on long-term changes in the peatland or on using the peat as an archive of regional climate change (e.g. Milner et al., 2016; Blundell et al., 2018). Instead, we use a fine-scale palaeoecological approach that allows us to understand recent peatland behaviour and regime shifts. The only similar study done on blanket peats that we are aware of is by Crowe et al. (2008) which provided useful insights into re-vegetation after erosion, but the cores were not dated, and the analysis focused mostly on vegetation changes rather than water table and C accumulation. Our work complements other fine scale palaeo studies such as Swindles et al. (2015) and Taylor et al. (2019) but differs from them in the type of peatland studied and the nature of the ecological dynamics (neither looked at recovery from erosion).

Recent carbon dynamics

Our data show that new peat is rapidly accumulating after the switch from the eroding state, while at the same time, older remnant peat is becoming protected from rapid aerobic decay by a higher water table. The timing of the switch to new C accumulation between the late 1800s and 1930s is earlier than other UK sites that see re-vegetation occurring from the 1950–70s CE onwards (e.g. the Pennines, UK – Evans and Warburton, 2007), although evidence of the onset of re-vegetation in these other areas is partly anecdotal (M. Evans pers. comm.). We have identified when re-vegetation first occurred in parts of our site, but the re-vegetating area is expanding and some areas of bare peat have yet to undergo a switch to the re-vegetated state (e.g. areas to the south of our core sites, Fig. 2). The transition from erosion to re-vegetation, and the accumulation of new C, is therefore an ongoing process. It can be misleading to compare rates of C addition, as new peat, with rates of net C accumulation reported from other studies because the latter includes decay losses from older, underlying peat. However, as we note above, the rates of C addition recorded for the two cores with no remnant old peat are equivalent to net C accumulation (MIG 1 and MIG4). Therefore, it is possible to compare our data with rates of net C accumulation from other studies. The UK site that most closely matches our study site is probably Moor House in the North Pennines of northern England. The site is generally higher than our site (450–893 m above sea level – Worrall et al. (2009)) but has a similar annual rainfall (1953 mm – Worrall et al. (2009)) and a similar grazing and pollution history as far as they are known. Worrall et al. (2009) produced estimates of the total C budget for the 11.4 km^2 Trout Beck catchment at Moor House between 1993 and 2005, and calculated an average net uptake of C of 56 g C m^{-2} yr^{-1}, with a range of 20–91 g C m^{-2} yr^{-1} for the 13 year period. MIG1 and MIG4 show net C accumulation rates about twice the average estimated by Worrall et al. (2009) and somewhat higher than the top of their range (i.e. ~110–120 g C m^{-2} yr^{-1}, see Results). Our high rates probably reflect ideal peat-forming conditions in the re-vegetating gullies. The estimates from Worrall et al. (2019) will include areas within the Trout Beck catchment where peat may still be undergoing degradation or where, because of the location (e.g. steeper hillslopes with deeper water tables) conditions for net peat accumulation are not optimal.

Worrall et al. (2019) estimated net C accumulation rates using contemporary flux measurements and empirical models derived from such measurements. Peat-core estimates of long-term rates of net C accumulation in northern peatlands are typically 22.9 ± 2.0 g C m^{-2} yr^{-1}, with higher rates of 24–32 g C m^{-2} yr^{-1} and up to 80 g C m^{-2} yr^{-1} over the past millennium (Loisel et al., 2014; Gallego-Sala et al., 2018). In our cores, the average C accumulation was between 46 and 121 g C m^{-2} yr^{-1}. Estimates of more recent apparent rates of C accumulation in northern peatlands are often higher, e.g. 51–149 g C

m^{-2} yr^{-1} (Piilo et al. 2019); however, these higher rates are, in part, due to an artefact known as the 'acrotelm effect'. Even under a constant climate, rates of C accumulation calculated for near-surface peat layers produce an artefact: an apparent increase (Clymo et al., 1998; Turner et al., 2014; Young et al., 2019). This is caused by the younger litter and peat near the peatland surface having undergone less decay and decomposition than deeper layers. Unfortunately, this effect is very difficult to correct for (Young et al., 2019).

The effects of changes to the climate and of industrial activity

Of the external factors that we considered, only two – climate change and quarrying – could plausibly explain our results, but even then, probably only partially. The timing of changes in climate and quarrying approximately coincide with the switch and rapid peat growth, but both factors require additional mechanisms to fully explain the data as we discuss below. This led us to consider whether a loss of resilience of the eroding state, due to changes in the way rainwater collects and flows through the erosion gullies, might also offer an explanation for the switch.

Our data meet the requirements of the definition of a shift to a different regime or state given in the introduction; i.e., they suggest a wholesale change in how the system functions (it is 'qualitatively different' – see Lenton (2013)). In some peatlands, year-on-year changes between C accumulation and loss may occur (e.g. Roulet et al., 2007) but these reflect an essentially stable system responding to variability in driving (boundary) conditions (e.g. meteorological), and, possibly, internal system noise (stochasticity – see Lenton (2013)). In contrast, at our study site we see a fundamental change in how the peatland behaves, from a condition of long-term C loss, with bare eroding surfaces and a patchy vegetation cover vulnerable to 'wash out' (see below), to one with a spatially-continuous vegetation cover, higher water tables and sustained (year-on-year) peat and C accumulation.

At large spatial scales, climate is the dominant control on peatland initiation and peat accumulation (Charman et al., 2013; Morris et al., 2018). For example, a widespread increase in C accumulation rates over the past millennium has been attributed to increased plant productivity linked to climate warming (Yu et al., 2009; Charman et al*.,* 2013; Loisel et al., 2014; Gallego-Sala et al., 2018; Zhang et al., 2018; Piilo et al., 2019). However, simply matching an increase in rates of C accumulation with recent changes in climate can be misleading because of the acrotelm effect noted above. Therefore, while it is possible that a warming climate from the middle of the 19th century to the present has provided favourable conditions for plant growth, leading to plant productivity exceeding decomposition, we cannot unequivocally attribute the switch and period of rapid peat accumulation to this amelioration. In addition, especially at smaller-scales, for peat to accumulate, site-specific factors need to be suitable (e.g. local topographic conditions creating suitable hydrological conditions (Zhang et al., 2018)) and peat accumulation at the site-scale cannot therefore be explained solely by climate.

It is also possible that mining activities in the 19th and 20th centuries provided a mild fertilisation effect and increased plant productivity, contributing to the switch to peat accumulation. As we note above, mining activity in the region around the field site coincided with re-vegetation and peat accumulation, and an increase in Fe and K concentrations in the peat. Mining dust can increase the base mineral input and nutrient supply to peatlands (Ireland et al., 2014; Gałka et al., 2019), and there is evidence of high C accumulation rates related to dust supply (Kylander et al., 2018). On bare, eroding peat surfaces, input of mineral-rich dust can promote growth of minerotrophic wetland plant species (e.g. *Juncus*, *Carex*) or species with high nutrient-use efficiency (e.g. *Eriophorum vaginatum* (McGraw and Chapin 1989)), both of which can stabilise the peat surface for *Sphagnum* to re-establish (*cf*. Paal et al., 2009). Indeed, peatland restoration programmes often include fertilisation of bare peat to encourage nurse species to re-establish (e.g. Yorkshire Peat Partnership, n.d.; Alderson et al. 2019). The successional pathways at the study site show initial colonisation by *E. vaginatum* and *J. effusus*, which may have acted as nurse species before *S. fallax* dominates (Fig. 3). However, given that the erosion had exposed the mineral ground in places, it is unlikely that external nutrient enrichment would be needed for these vascular plants to establish. Continued dust input through mining activities would not be conducive to *Sphagnum* growth because particulate pollution slows growth through a range of physiological and morphological responses (Farmer, 1993; Ireland et al., 2014). However, *S. fallax* – the dominant *Sphagnum* species at the site – is a common species in drained and damaged peatlands (e.g. in Central Europe (Gałka and Lamentowicz, 2014; Marcisz et al., 2015; Simova et al., 2019)) and there is evidence that it can tolerate higher mineral levels, increased nutrient supply and some pollution (Kooijman and Kanne, 1993; Limpens et al., 2003); and mining activity in the region slowly declined from the turn of the 20th century. Changes in climate and quarrying, although probably implicated in the switch, do not therefore provide simple and conclusive explanations for our findings.

The loss of resilience of the eroding state caused by changes in the landscape and its hydrological behaviour

In addition to climate and quarrying, internal ecosystem processes might explain why the change in system state occurred. After gullies first formed, the erosion state would have been maintained because rainwater, concentrated by narrow gullies, would have eroded peat from the gully floors and sides and 'washed out' any vegetation trying to establish there. Over time, however, gullies would have widened until reaching a critical width, after which rainwater was no longer sufficiently concentrated to wash out new, peat-forming vegetation. Our data do not allow us to say for certain if the shift between states was discontinuous, as is the case in systems exhibiting bi-stability or multiple stable states (e.g. Lenton, 2013; Petraitis, 2013), or whether it occurred as a continuous change in a monostable system (e.g. Scheffer et al., 2009; Lenton, 2013; Petraitis, 2013). However, as discussed below, the shift to a new state is consistent with a type of tipping point called a bifurcation. To paraphrase Lenton (2013), a bifurcation may be defined as the situation where a small, 'smooth', change to a control parameter – the width of an erosion gully in our case – causes a sudden qualitative or topological change to a system's behaviour (i.e., a fundamental change in its functioning). As the system approaches a bifurcation tipping point, it may exhibit critical slowing down (recovery from perturbation takes longer) and a greater variance in its properties (known as 'flickering') (Lenton, 2013). These indicators can act as an early warning that a system is approaching a tipping point (Scheffer et al., 2009).

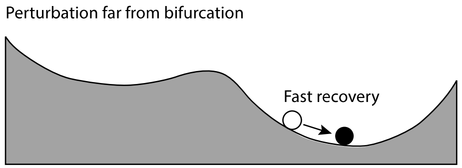
In Fig. 4 we envisage the study site undergoing morphological changes that reduce the resilience of the eroding state to perturbation (see below) until the switch occurs. We also propose the self-amplifying (positive feedback) mechanisms responsible for the change to renewed peat and C accumulation. In the left-hand column of Fig. 4, we adapt a figure from Lenton (2013) that uses the analogy of a ball in a 'stability landscape' to illustrate how the system moves towards a bifurcation-type tipping point. The *x* axis on the landscape represents the system state, which initially has two depressions or bowls, also called 'potential wells'. Although a continuum of states or positions along the x axis is possible, the ball (representing the state of the system at any particular time) tends to settle in the base of a bowl, which may be considered an attractor. The two attractors here are the two states being considered (erosion as the right-hand potential well, and re-vegetation combined with peat accumulation as the well on the left). As erosion gullies in the real landscape deepen and widen, the erosion attractor weakens, shown by its potential well in the stability landscape becoming shallower and slightly narrower (i.e., it is becoming less resilient to perturbation). Eventually, the system reaches a point where the potential well representing the erosion state disappears and the ball moves to the alternative attractor representing the re-vegetated, peat-accumulating state.

As noted in Fig. 4, perturbations are periods of *lower-intensity rainfall*. The erosion state is maintained by vegetation being washed out. During periods where less intense rainfall occurs, vegetation may establish from seedbanks or plant fragments originating from bare peat or neighbouring intact peatland, moving the ball in the direction of the left-hand state in Fig. 4 (peat accumulation state); a perturbation occurs. After the resumption of more intensive rainfall events, vegetation is removed and erosion resumes. However, after the gully reaches a critical width, a similar-sized perturbation may cause the system to switch to the new state (Stage 4: peat accumulation). Periods of lower-intensity rainfall can occur in a system that is essentially static climatically; i.e., they may simply represent variability within a given climate.

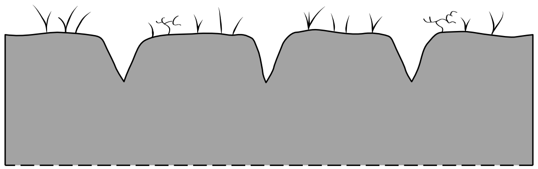
In Fig. 4 we consider how the switch might occur in gullies that have eroded down to a more erosion-resistant mineral substrate. In two of our cores – MIG2 and MIG3 – the switch occurred before the mineral substrate had been reached, which suggests that other tipping-point mechanisms were involved in these cases. However, gullies may become wider before an erosion-resistant layer has been reached. In peatlands on shallow slopes, such as our study site, lateral retreat of gully walls occurs via slumping and block failure (mass movements) and processes such as wind erosion, frost heave and rain splash erosion. Peat derived from these processes is deposited in the gully base and removed by running water (Evans and Warburton, 2007).

The stages described in Fig. 4 essentially use tipping-point theory to formalise a similar idea presented by Wishart and Warburton (2001), with one important and substantial difference. In their study of eroding blanket peatlands in the Cheviot Hills of southern Scotland, they envisaged the system becoming stuck at Stage 3; they suggested that channels or small streams in the widening gully would periodically erode newly-established vegetation and continue to undercut the gully walls until all of the original peatland had been eroded away. They also note that there are no feedback mechanisms in their conceptual model that see the system return to a state of "undisturbed blanket peat" (Wishart and Warburton, 2001). This view contrasts with our conceptual model where the streams disappear, and infilling becomes self-reinforcing. Our model fits observations at our study site. We have visited all of the re-vegetating gullies at our site and most small channels or streams, where they occur, have become clogged with vegetation; i.e., they do not conform to the scenario depicted by Wishart and Warburton (2001). More recently, Evans and Warburton (2007) hypothesised a sequence essentially the same as depicted in Fig. 4. They propose a conceptual "cut and fill" model of cycling between erosion and re-vegetation in blanket peatlands. In their model, they also consider other re-vegetation mechanisms such as natural damming of gullies by blocks of eroding peat falling from the gully sides.

Stability landscape 1

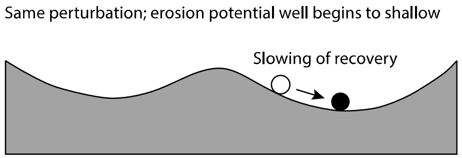


Stage 1. Rapid deepening of newly formed gullies

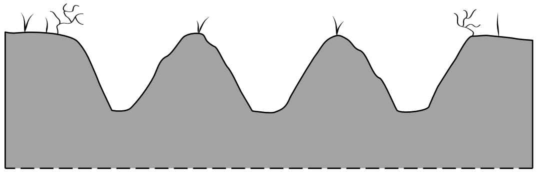


Water flows within gullies during wide range of rainfall events (small to large storms); flow is highly concentrated. Gully sides and bases actively eroding and too unstable for vegetation establishment. Periods of low rainfall (perturbations) in which erosion does not occur are short-lived (weeks) and system soon returns to state of active erosion during periods of abundant rainfall. Below: pictorial representation of the real system, showing incision of the peat in newly-formed gullies. Dashed line represents boundary between peat and underlying mineral material.

Stability landscape 2

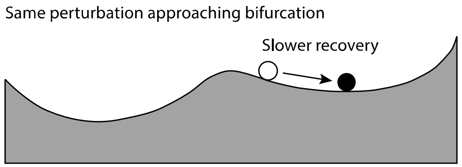


Stage 2. Gully deepening and widening

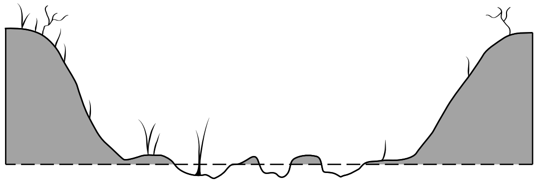


Because of widening of gullies, powerful, concentrated flows no longer occur during low-intensity rainfall; however, concentrated flows continue to occur during mid- to high-intensity rainstorms. Gully sides and bases actively eroding. Gully bases occasionally support vegetation for short periods (~1–3 years) during which, by chance, only low-intensity rainstorms occur. On return of mid- and high-intensity rains, powerful flows in the gully bases strip or 'wash out' the vegetation on the gully floors.

Stability landscape 3

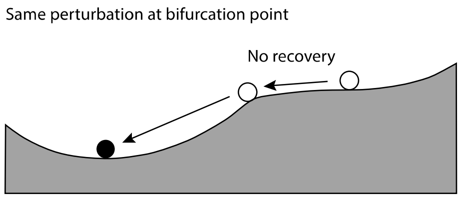


Stage 3. Gully widening/amalgamation

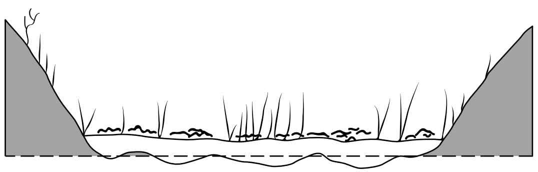


Gully base reaches erosion-resistant mineral substrate. Small channels or streams may form in the mineral material and migrate at times to the gully sides where erosion may still occur. However, in general, as gully widens, flows tend to become more diffuse across the gully base allowing vegetation to establish. With continued gully widening, only the rarest of rainstorms produce powerful enough flows to strip vegetation off the gully floor; system takes longer and longer to return to a state of active erosion, and the erosion potential well (see left) steadily shallows. System exhibits critical slowing down behaviour, with greater variance in system behaviour: it 'flickers' between being vegetated and bare/eroding.

Stability landscape 4



Stage 4. Gully stabilisation and re-vegetation



Gully widens further and, as plants become established, they provide friction to flow, reducing flow velocities and the chance of washout, thus setting in train a positive feedback whereby more plants can establish. Early successional plant species such as *Juncus* also alter the microclimate (act as a nurse species – see review in Evans and Warburton, 2007), allowing *Sphagnum* spp. and other bryophytes to establish. These in turn enhance peat accumulation, such that more water flow can be accommodated below the ground surface, reducing further any damaging surface flows. System has undergone a regime shift.

Figure 4. The weakening of the resilience of the peatland as it moves towards a bifurcation-type tipping point. The peatland is represented by a theoretical stability landscape (following Lenton, 2013) in the left-hand column, and by a cartoon of a cross-section of peatland in the right-hand column

Based on the above discussion we propose that a non-reversible shift takes place between the two equilibrium C states (eroding and accumulating) found in gullies within hagged blanket peatlands (Fig. 5). As we have already noted, our downcore data do not allow us to say if this switch relates to a monostable or bistable system, but the system shifts states in a way that is consistent with a bifurcation tipping point, and our observations lead us to believe that the shift from C loss to C accumulation is not reversible. That is not to say that erosion cannot restart, but once the gully has begun to accumulate C it does not become progressively narrower, reversing the sequence shown in Fig. 4 (i.e. from Stage 4 to Stage 3 to Stage 2 to Stage 1). Instead, new vegetation accumulates across the width of the gully, effectively filling it in from the base up. For erosion to begin again, a new gully would have to form in the infilling material. In this case a shift to an eroding state would move directly from Stage 4 to Stage 1 (Fig. 4). Nevertheless, until a gully completely infills, it will receive water from higher, uneroded, areas between gullies; i.e., it will remain a site of flow concentration. For this reason, it may be more prone to new gully formation than the uneroded areas, and this possibility would tend to support the cyclical model of Evans and Warburton (2007).

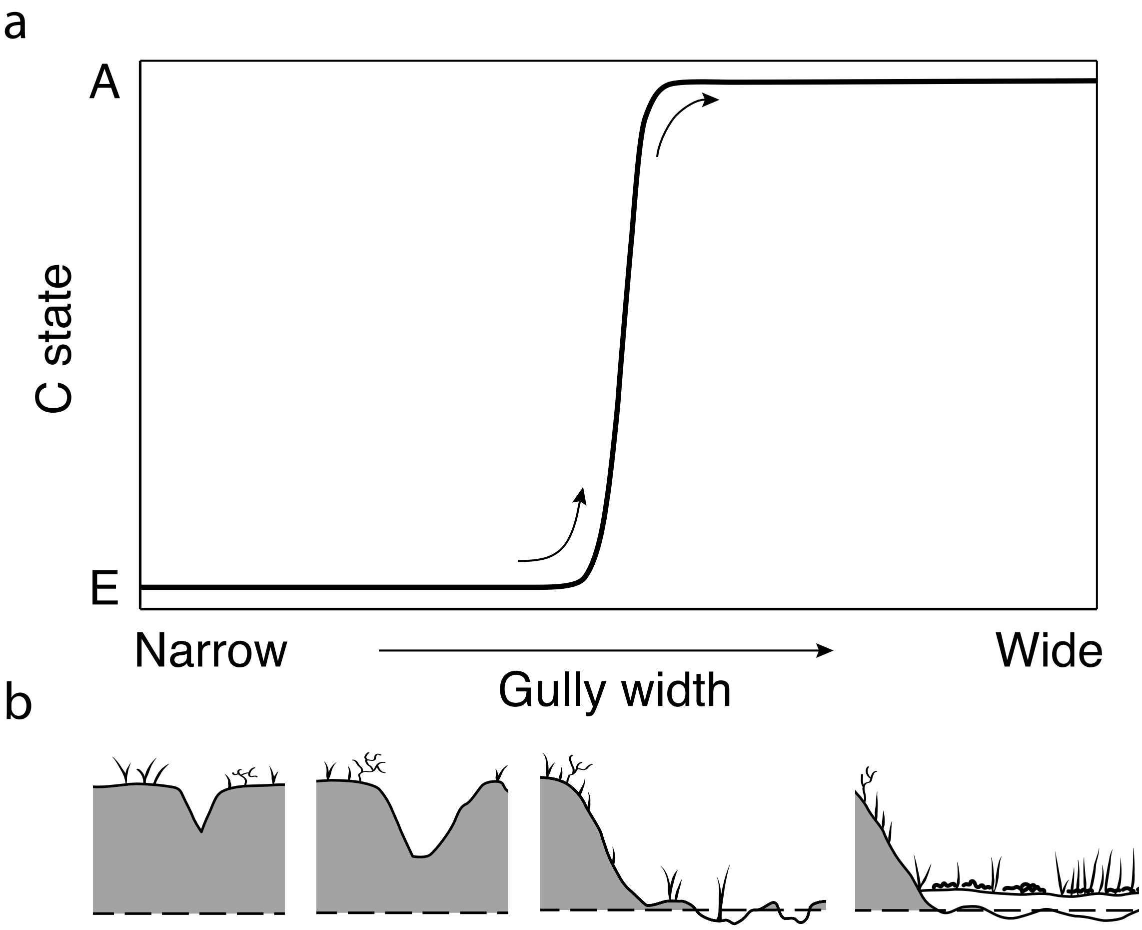


Figure 5. Shift from an eroding to an accumulating C state with a change in gully width. a) Conceptual model of the shift in C accumulation state from eroding (E) to accumulating (A). Arrows indicate a non-reversible shift between equilibrium states (refer to text). b) Cross sections of peatland corresponding to the C accumulation states (taken from Fig. 4).

Our model of events in Figs 4 and 5 shows how the shift is driven by internal morphological changes in the system under a stable climatic forcing. However, changes in state are also affected by boundary conditions. For example, climatic changes may combine with internal morphological changes to cause a shift. If the climate warms and dries, vegetation may grow more rapidly on the floor of the gully during periods with less intensive rainfall (such as the periods of low rainfall identified across England and Wales in recent centuries – see SI), and damaging flows of water will be less common because of the drier climate, meaning that a state shift may occur for a narrower gully than would be the case if the climate did not change. Therefore, it is possible for the changes in climate that we discuss earlier to modify the internal mechanisms causing a regime change. Finally, our model does not preclude switches occurring at different times across a landscape. Gullies vary in width and stage of development within a peatland (including along individual gullies), and this variability can be expected to be reflected in differences in the timing of switches from the eroding to accumulating states as observed in our cores.

Non-linearity and thresholds in peatland behaviour

Our study provides a clear example of threshold behaviour in a peatland system, and points to an additional way in which we might investigate peatlands and conceptualise their behaviour. Explicitly thinking in terms of a tipping point framework helps reveal possible internal mechanisms by which sudden and fundamental changes in system functioning may occur. Studies such as that of Charman et al. (2013), who suggest that northern peatlands may become stronger C sinks under a warming climate, show that peatlands can display essentially linear behaviour over centennial timescales, and it is tempting to assume that such linear behaviour will continue into the future as the climate continues to warm. However, as we show above, internal 'evolution' of a peatland can bring about fundamental, step-like, changes in peatland behaviour. In this context, it is notable that peatland development models such as DigiBog (Baird et al., 2011) and HPM (Frolking et al., 2010) tend to include strong negative feedbacks (stability within a single state). For example, as shown in the modelling study by Swindles et al. (2012), a drying of the climate causes an initial deepening of water tables in a peatland but also causes a reduction in peat permeability (hydraulic conductivity) through aerobic decay that reduces rates of water flow from the peatland, so that the water-table position relative to the peatland surface returns to its pre-drying position. While such strong negative feedbacks certainly occur in peatlands, and perhaps even dominate (see Belyea, 2009), other possibilities exist and are worth considering. For example, under future climates, with an increase in drought frequency and duration, a peatland may start to crack. Although the peat between cracks may decay and undergo a reduction in its permeability, the effect of cracks as conduits for rapid water flow may override this reduction, causing water tables to fall further below the surface, to a new deep water-table state, that could in turn lead to changes in peatland vegetation and its C balance. In other words, threshold behaviour may occur, and peatlands may undergo topological changes in state as the climate warms.

Management implications

We have shown how peatlands can 'self-repair' from an eroding state, both in terms of their vegetation and hydrological functioning. Eroded blanket peatlands are priority sites for conservation in the UK (IUCN, 2018), and efforts are often made to reduce or halt erosion by placing dams across gullies (e.g., Armstrong et al. 2009; Parry et al. 2014; MoorLIFE 2020). Our tipping point model (Fig. 4) shows why such efforts are generally successful but also points to alternative interventions that might be considered. A series of dams along a gully will reduce water-level gradients in the stretches between the dams. Flow will deepen but flow velocities will be reduced, lowering the probability of washout of any vegetation that may establish. Damming gullies would not generally change flow discharge along them, because that is controlled by upstream catchment area and net rainfall. Although effective at reducing erosion or stimulating re-vegetation, a series of dams may be undesirable because of erosion downstream of a dam caused by a plunge pool, although measures may be taken to prevent this (e.g. using permeable dam materials that decrease flow velocities rather than create a watertight seal (Armstrong et al., 2009)). Pools also tend to form behind dams and these may be undesirable because of the higher CH_4 emissions associated with them (e.g., Waddington and Roulet, 1996; Laine et al., 2007). An alternative to damming is to widen gullies artificially so that a state transition to re-vegetation and C accumulation is more likely, or to reduce water flow down a gully by re-routing water flow pathways in its upper catchment. Both alternatives require re-grading of the peatland landscape using mechanical diggers, but such large-scale works already take place in some degraded UK peatlands (Yorkshire Peat Partnership, n.d.). Of course, another alternative is to do nothing, especially if the peatland looks like it is close to a tipping point already (e.g., a partially vegetated gully containing patches of new peat). Through reconstructing a trajectory of peatland self-recovery, our results provide an additional perspective on management options for eroding blanket peatlands

Conclusions

Much of the literature on peatlands is focused on how peatlands become degraded and switch from C sinks to sources. Similarly, much of the literature on tipping points focusses on shifts from a ‘good’ to a ‘bad’ state. Unusually, we find the opposite and set out to understand what causes a peatland to switch from active erosion with exposed peat, loss of vegetation and loss of C, to re-vegetation, higher water tables and renewed C accumulation over short timescales. Between 5.2 and 10.6 kg m^{-2} of new C has accumulated since the switch, occurring between the late 1800s and early to mid 1900s, with average C accumulation rates as new peat between 46 and 121 g C m^{-2} yr^{-1}. Our palaeoecological data reveal a switch in system functioning that can be explained as a bifurcation tipping point. External factors, such as climate and pollution levels, are likely to be important for setting suitable boundary conditions to shift a peatland towards recovery, but internal mechanisms (i.e. evolution of gully network) also offer an explanation for the changes we observe in the system's functioning. Of course, recovery may not last as the climate changes, and new types of degradation may occur such as the cracking of peat mentioned above. Nevertheless, for both recovery and degradation, we show how application of tipping point theory can help improve understanding of some important aspects of peatland behaviour.

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Authors' contributions

A.M.M and A.J.B conceived the idea, designed the methodology and led the fieldwork and data collection. S.M.G, G.T.S, N.S, M.S.I.T and M.G. collected and analysed data. D.M.Y provided critical input for the discussion, in particular the part on tipping-point theory. A.M.M and A.J.B led the writing of the manuscript. All authors contributed to drafting the manuscript and gave final approval for publication.

Data Availability Statement

The peat core data, including the palaeoecological and peat properties data, are available in the Dryad Digital Repository (Milner et al., 2020 doi: 10.5061/dryad.sbcc2fr3r). Other supporting information is available in the online supporting information.

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