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Trajectories of ecosystem change in restored blanket peatlands

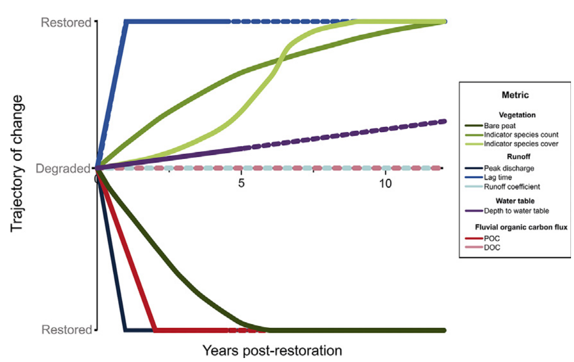
Danielle M. Alderson [a,*], Martin G. Evans [a], Emma L. Shuttleworth [a], Michael Pilkington [b], Tom Spencer [b], Jonathan Walker [b,1], Timothy E.H. Allott [a]

[a] Upland Environments Research Unit, Geography, University of Manchester, United Kingdom of Great Britain and Northern Ireland
[b] Moors for the Future Partnership, United Kingdom of Great Britain and Northern Ireland

HIGHLIGHTS

- Ecosystem services show phased recovery following restoration in eroded peatlands.
- Re-vegetation is key to reporting project success over short funding timeframes.
- Results support wide-scale adoption of Lime-Seed-Fertiliser-Mulch restoration.

GRAPHICAL ABSTRACT



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ABSTRACT

Peatlands provide a range of ecosystem services but are sensitive to changes in climate and land-use, and many peatlands globally are degraded. We analyse a large-scale, unique and diverse dataset, collected over 15 years, as part of major landscape scale blanket peat restoration projects in the south Pennines, UK. Trajectories of ecosystem change after restoration were assessed by measuring key ecosystem responses including: vegetation cover and indicator species; water table, runoff, and water quality.

The reactions of these metrics vary in their behaviour, both in the timing of first response and the rate of change towards a new stable state. Re-establishment of vegetation is a key driver in rapidly reducing particulate carbon loss and attenuating stormflow, while over time biodiversity is improved by the return of native species, and water tables gradually rise. The phasing of these ecosystem service responses indicates that there are different characteristic timescales for the improvement of peatland functions, driven by both surface and subsurface processes. The rapid establishment of vegetation cover over two years, and its importance in improving a broad range of functions, signify it as a key milestone for reporting project success within typical funding timeframes. This study supports the adoption of Lime-Fertiliser-Seed-Mulch restoration on eroding blanket peatlands globally. The trajectories developed are important to help guide practitioners of ecological restoration. Better understanding of the dynamics underpinning the slower response times of subsurface hydrological and biogeochemical function is identified as a key knowledge gap. An understanding of the limits of ecosystems recovery is important when target setting for restoration projects, especially where attaining pristine conditions is unachievable.

* Corresponding author at: Researchers Suite, 1st Floor, Arthur Lewis Building, Geography, School of Environment, Education and Development, The University of Manchester, Manchester M13 9PL, United Kingdom of Great Britain and Northern Ireland.
E-mail address: danielle.alderson@manchester.ac.uk (D.M. Alderson).
[1] Present affiliation: Swansea University, United Kingdom of Great Britain and Northern Ireland.

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1. Introduction

Restoration ecology is centred around restoring degraded, damaged or destroyed ecosystems and has a longstanding interest in understanding trajectories of ecological change in restoration projects. Defining the probable trajectory of a restoration intervention allows simple definition of success criteria, and of suitable timescales for restoration monitoring (Suding and Gross, 2013). Ecological theory predicts a variety of characteristic forms of restoration trajectory, but real world experience suggests that in many cases restoration does not make a smooth transition from degraded conditions to a pristine ecosystem state (Bullock et al., 2011; Cortina et al., 2006; Matthews et al., 2009; Zedler and Callaway, 1999). Mismatches between desired changes in ecosystem state based on ecological theory, and observed practical restoration outcome, may be due to changes in the physical boundary conditions of the system prior to, and during the restoration period. Assessing the full response to restoration intervention therefore requires understanding of trajectories of change in the physical system, as well as the ecological one. This focus is also relevant as restoration aims are increasingly couched not only in terms of improving biodiversity, but also in terms of delivering specified ecosystem services (Bullock et al., 2011; Choi, 2007), which may include regulation of physical systems such as runoff or erosion.

Peatlands provide a wide range of ecosystem services, and are most commonly known for their crucial role in global climate regulation, by storing >30% of total global soil carbon (Bonn et al., 2016). Peatlands are vulnerable systems, sensitive to subtle changes in climate, land use and pollution (Evans et al., 2014). As a consequence of human intervention in these environments, peatlands across Europe are ubiquitously degraded (Joosten, 2009) and the previous intact status of boreal and tropical peatlands is at risk, as a result of changing land use (e.g. Page et al., 2008). Degraded peatland systems have been a focus of considerable work on approaches to ecosystem service restoration (Bonn et al., 2016), and on trajectories of change in restored systems. A range of well-established restoration approaches have been developed, including drain blocking (Armstrong et al., 2009); tree removal (Anderson et al., 2016); topsoil removal and diaspore transfer for fen restoration (Klimkowska et al., 2007); moss layer transfer to restore mined peatlands (Rochefort et al., 2003); and revegetation and gully blocking (Parry et al., 2014). This paper reports for the first time on trajectories of ecosystem services change, in relation to peatland restoration approaches which have been developed for the severely eroded blanket peatlands of the UK.

The blanket peatlands of the UK and Ireland are globally distinctive in the degree and intensity of physical erosion that they have suffered in the last millennium (Evans and Warburton, 2007). More recently, these systems have been sites of extensive landscape scale restoration. In the UK, peatlands are important carbon stores containing 2.3 Mt C (Billett et al., 2010) and 96% of peatland area is blanket peatland (JNCC, 2011). Blanket peatlands also support a range of other ecosystem services, including provision of clean water, runoff regulation, sediment control, recreation and cultural services, and extensive agriculture (Bonn et al., 2016). However, while intact peatlands are significant carbon sinks, erosion can lead to the transformation of areas of peat from carbon stores to carbon sources (Evans and Lindsay, 2010; Worrall et al., 2009), and can significantly impact other ecosystem services such as floodwater regulation and sediment control.

The range of ecosystem services provided by intact peatlands and the severity of their degradation, have meant that there has been a long history of development of restoration approaches. Restoration success in these environments typically aims to restore hydrological function, vegetation cover and in particular, active peat forming vegetation (Anderson et al., 2009). The success of rewetting is measured by assessing water tables and their proximity to the surface, whereas biodiversity is assessed by vegetation surveys including the assessment of recovery of keystone species (Buckler et al., 2013). Water table and vegetation cover are therefore arguably the two principle determinants of peatland ecosystem function. Many restoration monitoring programmes only monitor these two parameters, on the basis that they provide a metric of ecosystem change which might be taken as a proxy for wider change in ecosystem service provision (Holden et al., 2016). However, this paper will go beyond the measurement of these parameters, analysing other indicators of a functioning peatland system.

Analysis of the trajectories of change for metrics which may infer ecosystem service recovery post-restoration, requires extensive monitoring in time and space. In this paper we collate and analyse a unique dataset from a variety of separate projects, which has been collected alongside major landscape scale restorations in the south Pennines region, to develop trajectories of ecosystem change associated with the restoration of degraded blanket peatlands. Thus far, short-term changes of specific ecosystem functions have been analysed, on a project specific basis on occasion utilizing a BACI (Before After Control Impact) approach (Pilkington et al., 2016; Evans and Shuttleworth, 2016; Allott et al., 2015; Spencer and Evans, 2016; Pilkington and Crouch, 2015; Shuttleworth et al., 2015). This paper will go further, producing longer-term trajectories (in excess of 10 years) and considering multiple ecosystem service provisions and their interactions.

The aims of this paper are threefold: (i) To analyse a major dataset on peatlands to develop empirically informed trajectories of ecosystem change following the restoration of eroding blanket peat; (ii) To explore the temporal sequencing of changes in ecosystem function to inform process based understanding of blanket peat restoration; (iii) To consider the implications of the trajectories developed in this study to refine restoration work and future monitoring programmes. This will be achieved by synthesising data from multiple sources where commonalities allow analysis, to produce trajectories for metrics representative of a variety of ecosystem functions. We critically evaluate the utility of consolidating diverse datasets before producing a conceptual model of ecosystem recovery.

2. Methods and study sites

2.1. Study sites

This paper is based on data collected at five sites which have been restored and monitored at various time periods over the last 15 years, mainly within the Peak District National Park at the southern end of the Pennines range extending across northern England. Initial experimental work carried out in the severely eroded peatlands (Anderson et al., 1997), has led to the development of a standard approach to the re-vegetation of bare peat sites (Anderson et al., 2009). In this paper the method is referred to as the Lime-Seed-Fertiliser-Mulch (LSFM) approach. The LSFM approach involves spreading lime and fertiliser from the air at a landscape scale and aerial seeding of a nurse grass crop. The bare peat surfaces are typically mulched with cut heather. The lime, fertiliser and mulch create conditions for the rapid germination of the nurse crop so that it is sufficiently established to resist frost heave in the first winter. The aim of this initial stabilisation approach is not to establish non-native grasses, but to stabilise the bare peat surfaces, to facilitate growth of native moorland species. Alongside the revegetation works, selected gullies are blocked using an appropriate material such as loose stone, timber or heather bales. In the south Pennines alone, over 2500 ha of peatland have been restored in the last 12 years (Moors for the Future Partnership, 2015) and the approach is being applied and modified for use widely across the UK uplands.

Each main site was divided into a number of micro-catchments and experimental plots, variable between locations and for individual projects. These are locations where bare peat restoration has been undertaken using the restoration approaches described above. Fig. 1 shows key locations and site characteristics, while restoration treatments are summarised in Table 1. All sites are of a similar elevation, have comparable climatic conditions (within blanket bog formation criteria; Lindsay et al., 1988) and have been exposed to severe erosion.

2.2. Data collection and treatment

All the sites considered were subject to the LSFM method of re-vegetation. At some sites gullies were blocked at the same time as re-vegetation, but vegetation and water table data were not monitored on gully floors or gully edges so that what is presented here is likely to mainly represent the impact of the LSFM approach on the peat mass. The intention of this analysis is not to compare restoration methods, but the integrated impact of standard restoration approaches across the sites.

Four main ecosystem functions (vegetation cover, depth to water table, runoff response and fluvial organic carbon flux) were assessed using a number of different datasets. These metrics represent key ecosystem services that intact and functioning peatlands may provide. The parameters are inherently interlinked, and it is difficult to assign one metric to a single ecosystem service. High water tables in particular are linked to a variety of ecosystem services. Generally, these metrics are representative of ecosystem services such as biodiversity, runoff regulation and provision of clean water/sediment control.

Vegetation cover and water table were measured at all the sites, while runoff monitoring was restricted to Kinder Scout and carbon flux data were collected at both Bleaklow and Kinder Scout. Consistent methods, within the constraints of original project aims, have been applied across the study sites, in order to assess the trajectories of these ecosystem functions post-restoration. Because the restoration projects span the last 12 years with different start dates, all data are presented as years post-restoration. Further site details are outlined in Table 2.

Where comparable data were available from several sites, the data have been consolidated to form a single trajectory, as inter-site comparison is not the objective of this study. Data are presented as deviations from control conditions. Comparing restored sites with eroded bare peat sites provides a better assessment of changes associated with the restoration process. Calculating the relative difference between $(\text{metric})_{\text{treatment}} - (\text{metric})_{\text{control}}$ enables noise from natural variation to be removed, to extract the restoration signal. In addition, data has been normalised to a baseline representing pre-restoration characteristics, to measure real change associated with restorative practices. Positive values therefore indicate that the metric of interest is greater at the treatment site than at the bare control, while negative values indicate the opposite.

2.2.1. Vegetation data

Vegetation data were collected across 21 sub-sites (including bare peat controls and treatment sites) across all of the main sites during the period 2003–2015. Data does not exist ubiquitously for all sites across the time period specified as it was related to different projects, as such the length of time post-restoration varied at different sites. Vegetation surveys were completed using both random and stratified

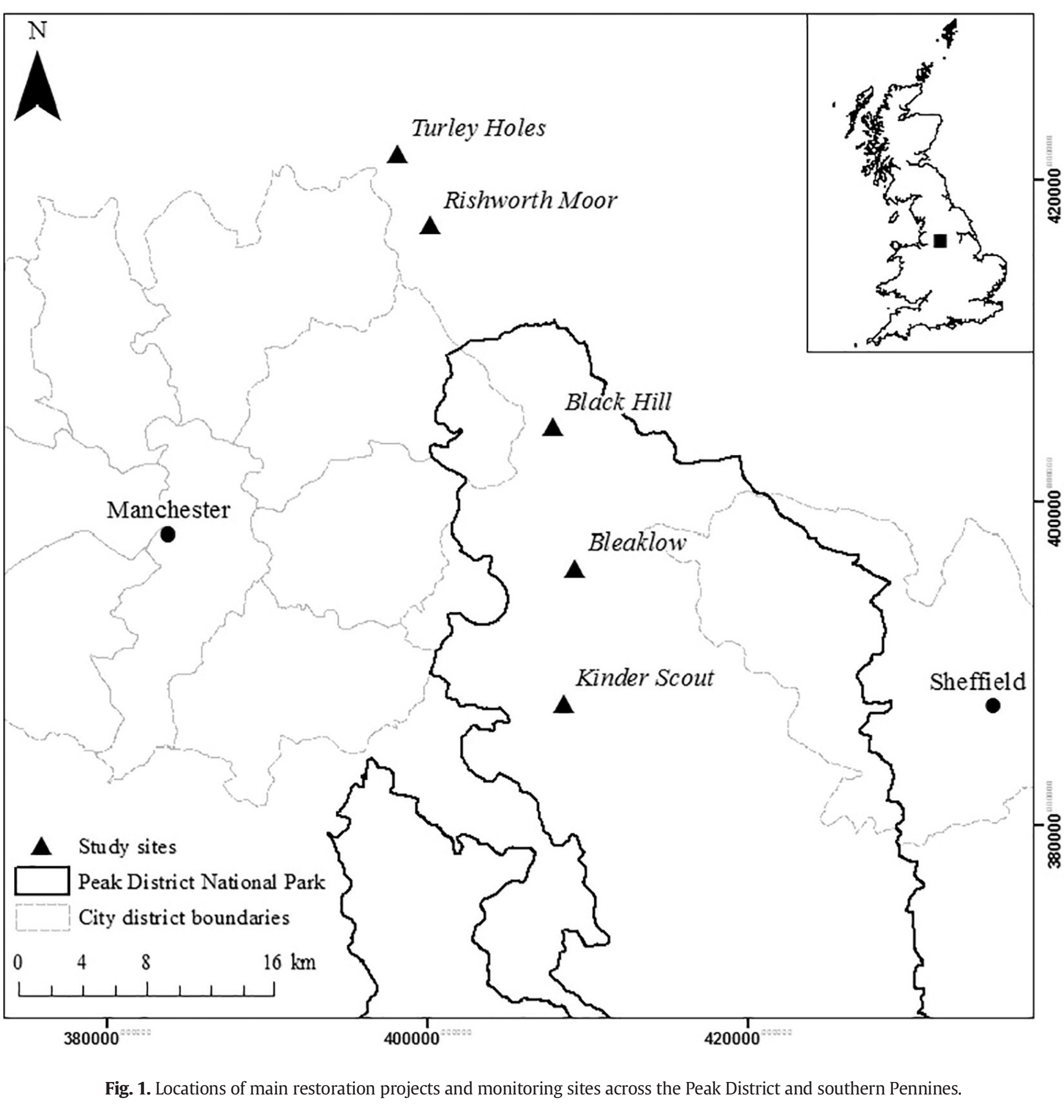

Fig. 1. Locations of main restoration projects and monitoring sites across the Peak District and southern Pennines.

Table 1
Characteristics and restoration treatments of key study sites.

Site	Specific project	Area	Maximum elevation (m)	Treatment date
Kinder Scout	DEFRA Making Space for Water	84 ha	636	2011–2013
Bleaklow	MoorLIFE	26 km^2	633	2003–2008 and 2011–2015
Rishworth Common	MoorLIFE	1727 ha	480	2011–2015
Turley Holes	MoorLIFE	665 ha	420	2011–2015
Black Hill	MoorLIFE 2020	46 ha	580	2005–2006, 2008, 2012–2015

placement of 2 m × 2 m quadrats, either before, or at the commencement of nurse crop seed application. As such, seeding was deemed as 'time zero'. Approximately 10 quadrats were used per site although this varied by site and year. The quadrats were analysed for three favourable condition attributes, selected as they were found to be the main cause for failure at MFFP sites (MFFP, pers comm) for meeting Natural England favourable condition standards (JNCC, 2009); percentage vegetation cover (measured as percentage bare peat), percentage cover of indicator species and indicator species richness. Indicator species for blanket bogs were derived from the JNCC (2009) Common Standards Monitoring Guidance. More information on procedure may be found in Pilkington et al. (2016). Data were available from all sites across the southern Pennines, but the final analysis presented here is based on Bleaklow and Kinder Scout as these were the only areas with associated control data.

Although extensive control data were collected at these sites, (Bleaklow: 2007–2014; Kinder Scout: 2010–2015), control data were not available for every year of treatment measurement, as some data collection began prior to 2007. Control sites were situated in areas of bare peat. Throughout the treatment period bare peat cover remained close to 100% (Interquartile range (IQR); Bleaklow: 97.12–99.13 and Kinder Scout: 98.41–99.74), indicator species counts remained close to zero (IQR; Bleaklow: 0.09–0.38 and Kinder Scout: 0.23–0.53), and indicator species cover also remained near zero as a consequence (IQR; Bleaklow: 0.09–1.10 and Kinder Scout: 0.11–0.21). For this reason, the differences in the value of these metrics between treatment and control for vegetated sites were calculated using median values across the years of measurement at the two respective control sites. Since the control data for indicator species metrics were close to zero, the difference relative to control values were in the vicinity of the raw measures of these metrics.

2.2.2. Water table data

Depth to water table data were measured at 17 sub-sites across all five areas, between 2010 and 2015, ranging in age post-restoration from 0 to 12 years. Experimental plots comprised a 30 × 30 m area, containing clusters of 15 dipwells (Allott et al., 2009). Dipwells (1 m length) made of polypropylene pipe (30 mm internal diameter) were drilled at 100 mm intervals creating perforation holes, allowing water levels to equilibrate. These dipwells were installed in boreholes of the same diameter, leaving 100 mm of pipe protruding above the ground surface. Manual measurements were taken from the clusters of dipwells weekly or bi-weekly between September and December using 10 concurrent measurements conducted manually, or with a purpose-constructed electronic dip-meter. Measurements were made relative to the ground surface. Detailed information on procedures can be found in Evans and Shuttleworth (2016).

2.2.3. Runoff data

Stormflow data were collected at two sites on Kinder Scout; a treatment site and a bare peat control. Runoff data were monitored from 2010 onwards. V-notch weirs and pressure inducers were installed, to record depth of water flowing over the v-notch weir at catchment outlets. This was converted to discharge, and standardised across sites by dividing by catchment area. Pressure transducers recorded flow depth at 10 min intervals, and tipping bucket rain gauges continuously monitored rainfall at the same interval. Storm hydrograph data were extracted using strict selection criteria (see Shuttleworth et al., 2019), resulting in storm events being comparable between sites. Three key metrics derived from this data were used to evaluate restoration trajectories; (i) peak storm discharge, (ii) lag time, and (iii) runoff coefficient. These metrics have been used successfully in previous work to demonstrate changes in runoff characteristics post-restoration (Shuttleworth et al., 2019), however the analysis here is completed on an extra year of data, strengthening the conclusions of the aforementioned study. Peak storm discharge is the maximum discharge during a storm event. Lag time is the interval between maximum rainfall intensity and peak storm discharge. Runoff coefficient is a measure of the proportion of storm precipitation that contributes to storm discharge.

2.2.4. Fluvial organic carbon data

Assessment of trajectories of fluvial organic carbon flux involved the collection of Dissolved Organic Carbon (DOC) and Particulate Organic Carbon (POC) data. DOC was monitored at Kinder Scout. On a fortnightly or weekly basis, 500 mL of water flowing over a v-notch weir was collected in a pre-rinsed water bottle. Samples were filtered at 0.45 μm and analysed for DOC using a Shimadzu TOC-V carbon analyzer. Full details of DOC collection procedure are within Spencer and Evans (2016).

POC datasets were collected from bare peat control and restored sites at Kinder Scout and Bleaklow. Modified Time-Integrated Mass Flux samplers (Shuttleworth et al., 2015) were installed in gullies. Five sampling campaigns of approximately 10 weeks were undertaken between October 2010 and January 2012. At the end of each campaign, sediment was extracted from the TIMS and stored at 4 °C prior to analysis. Samples were washed through an 8 mm sieve with deionised water, and dried at 40 °C. POC was determined by the Loss-On-Ignition method at 550 °C for 4 h. The OM content was converted to OC content. The same method was used at Kinder Scout, with the exception being that 10 traps were installed at each site during one campaign over 4 weeks. The mean mass of sediment trapped in the different traps

Table 2
Details on methods to assess key ecosystem functions.

Trajectory method	Site application	Notes	Data collection period
Vegetation cover	Black Hill, Bleaklow, Kinder Scout, Rishworth Moor, Turley Holes	$n = 21$ sites (18 treatment, 3 control)	2003–2015
Water table	Black Hill, Bleaklow, Kinder Scout, Rishworth Moor, Turley Holes	$n = 20$ sites (17 treatment, 3 control)	2010–2015
Runoff response	Kinder Scout	$n = 2$ sites (1 treatment, 1 control)	2010–2015
Fluvial organic carbon flux	DOC-Bleaklow and Kinder Scout	$n = 3$ sites (2 treatment, 1 control)	DOC-2011–2016
	POC-Bleaklow and Kinder Scout	$n = 5$ (3 treatment, 2 control)	POC-2010–2013

was taken, with the difference between control and treatment established as a percentage. At Kinder Scout the data were collected at 2 years post-restoration whereas the Bleaklow data were collected at 9 years post-restoration. Further details of data collection for POC flux can be found in Pilkington and Crouch (2015) for Kinder Scout and in Shuttleworth et al. (2015) for Bleaklow.

2.3. Statistical analysis

Data were collated from various sites for each year post-restoration where available. In many cases, the data were not normally distributed and required statistical treatment in a non-parametric manner. As such, for presentation of data and further statistical treatment, the median of datasets were used with the interquartile range utilised to account for inter-site variance. All data were normalised using baseline restoration data, so that the first year post-restoration was equivalent to time zero. Statistical analysis was carried out using IBM SPSS statistics version 22.

3. Results

3.1. Vegetation

Vegetation data were treated differently to the other ecosystem function data, as a consequence of the breadth of data across all geographical sites. Only two sites had associated control data, limiting the capacity for robust analysis. However the raw, untreated data may be used to look at general trajectories, while the more robust approach can be utilised to validate trends. As such, raw trajectories of change in vegetation characteristics over 12 years for all sites are presented in Fig. 2. The proportion of bare peat follows an asymptotic curve, decreasing rapidly from close to 100% at year 0, and approaching 25% cover after 2 years. The intersect between the trendline and the 95% CI of the asymptote constant of the regression equation provides a conservative estimate of the threshold after which there is little further change in cover - approximately 8 years (to the nearest year). Total indicator species count also follows an asymptotic curve, from no indicator species at year 0, accelerating rapidly during the first two years and subsequently reaching a plateau after 12 years post-restoration, (95% CI of the asymptote). Indicator species cover follows a dose response curve, starting at ~10% at year 0, increasing gradually during the first 3 years of recovery, before accelerating more rapidly, and reaching a new state approximately 9 years post-restoration, when considering the 95% CI of the asymptote.

Vegetation data from individual sites that had a control (Bleaklow and Kinder Scout) were compiled to produce a single median trajectory of vegetation cover and species richness (Fig. 3). This utilised the same analytical approach taken for the other metrics that have been analysed, enabling comparison. Bare peat cover follows an asymptotic curve with a rapid acceleration over 2 years, beginning to plateau after 8 years (95% CI of the asymptote). The trajectory of indicator species count is best captured by an asymptotic curve, following a gradual increase from year 0 and plateauing after 12 years post-restoration (95% CI of the asymptote). Indicator species cover follows a similar dose response curve with the $(\text{indicator species cover})_{\text{treatment}} - (\text{indicator species cover})_{\text{control}}$ data from all geographical areas, with a gradual increase in the first 2 years post-restoration, followed by a more rapid acceleration, achieving a new steady state after 9 years (95% CI of the asymptote).

Raw trajectory data presented here demonstrate that the trends found when applying the robust $(\text{metric})_{\text{treatment}} - (\text{metric})_{\text{control}}$ approach apply across a broader geographical area, strengthening the evidence for the trajectories observed. Most of the trajectories of vegetation characteristics are described by curvilinear or dose response trends. The curvilinear trends describe a rapid response to the LSFM measures, followed by a stabilisation generally over 9–12 years as bare peat is colonised by grass species and different indicator species compete, and some establish dominance. When taking into account the 12 years of data collection, indicator species count plateaus after 12 years of restoration. However continued data collection could improve this trajectory. Three key favourable condition attributes (JNCC, 2009), clearly demonstrate an immediate and rapid improvement before eventual stabilisation, following LSFM treatment, in comparison to control sites.

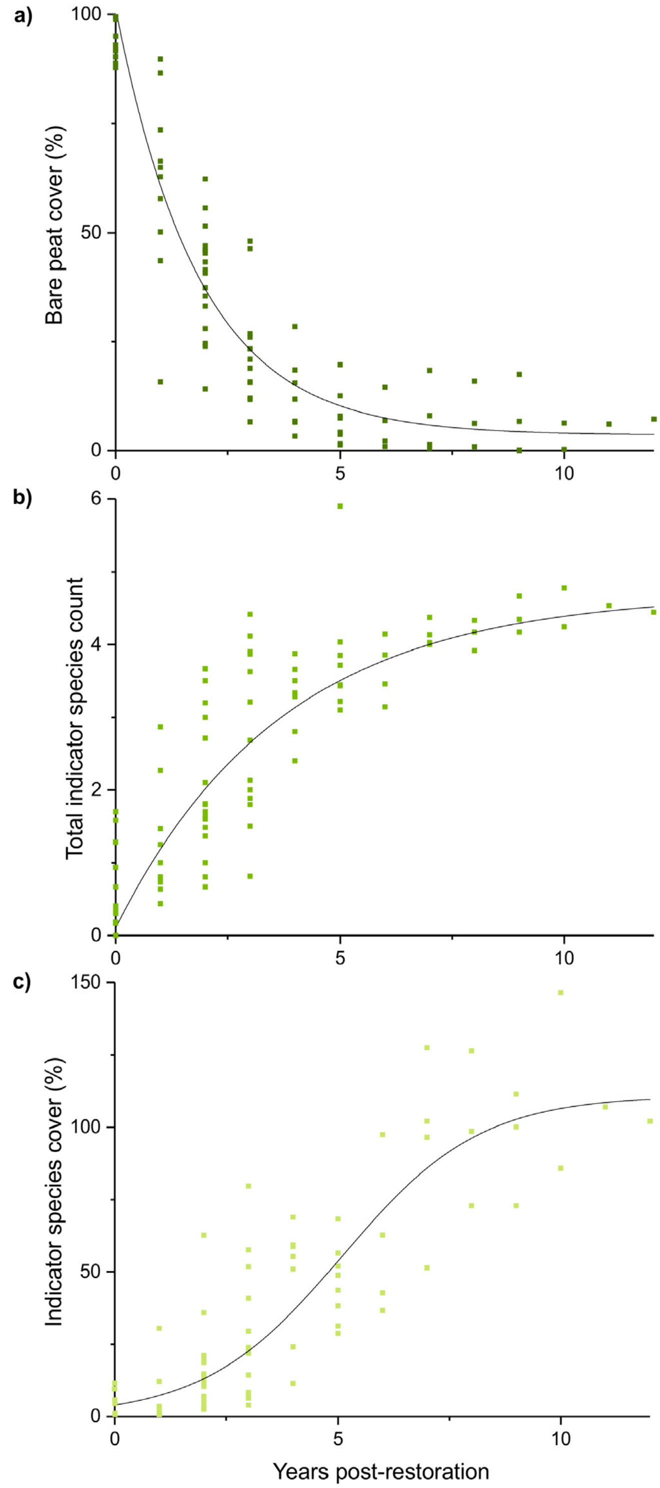

Fig. 2. Vegetation trajectories from all sites over 12 year restoration period a) median bare peat cover, b) median count of indicator species, c) median cover of indicator species.

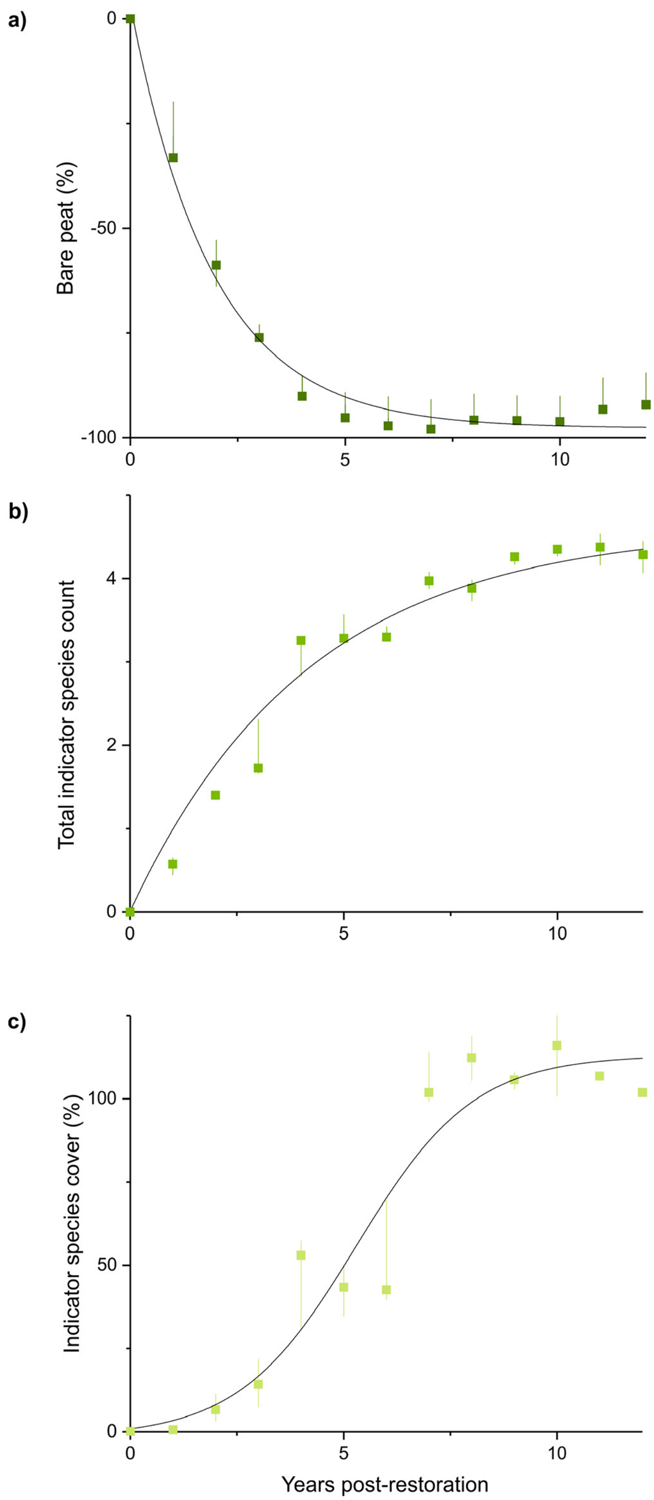

Fig. 3. Normalised median vegetation trajectories (treatment-control) for Bleaklow and Kinder Scout over 12 year post-restoration period. Errors defined as the interquartile range a) median bare peat cover, b) median count of indicator species, c) cover of indicator species.

3.2. Water table

Fig. 4 presents depth to water table data, derived from manual dipwell measurements, from seventeen locations, across the five study sites. Linear trajectories for depth to water table are present regardless of number of years post-restoration at different sites. Depth to water table data for the full 12 years of restoration were not available for any site. Instead linear trends were identified for different time windows for different sites, and combined in Fig. 4. Linear extrapolation was used to produce a median trajectory for all sites over the whole time period. Early stage data from recently restored sites were extrapolated to 12 years post-restoration, whereas late-stage data were extrapolated to the origin, assuming the whole period of restoration followed the linear trajectory in the latter stages of restoration. Variability between sites is expressed by the interquartile range.

The proximity of the water table to the surface increased (Fig. 4), amounting to an average of 8 mm yr^{-1} over the 12 years of surveying across all sites ($n = 17$). The distribution of data about the median fits more closely to the lower quartile, demonstrating that the upper quartile could potentially be a large overestimation of the depth to water table, based on forecasted data, or, alternatively, that there is less scope for negative residuals, as the water table is already at, or close to, the surface.

3.3. Runoff response

Fig. 5 presents runoff data from micro catchments on the Kinder Scout Plateau. Restoration had an immediate effect on two of the three runoff metrics. Peak storm discharge decreased at the treatment site relative to the control, while lag times increased. There was no obvious change in % runoff. There were no subsequent directional trends in the behaviour of any of the metrics following the pronounced step change in year 1. To demonstrate this, Kruskal-Wallis 1-way ANOVA were employed for each metric to look for differences/similarities between years. Groupings of statistically similar ($p > 0.05$) years are represented by lower case grey letters in Fig. 5.

Lag time shows the clearest evidence of a consistent step change in behaviour following restoration. Pre-restoration data fall into Group a, while all subsequent year's post-restoration falls into Group b, demonstrating that lag times were significantly different pre- and post-restoration and that lag times were statistically similar in the four years following restoration. The step change is slightly less clear for peak discharge. Three out of the four years post-restoration fall into

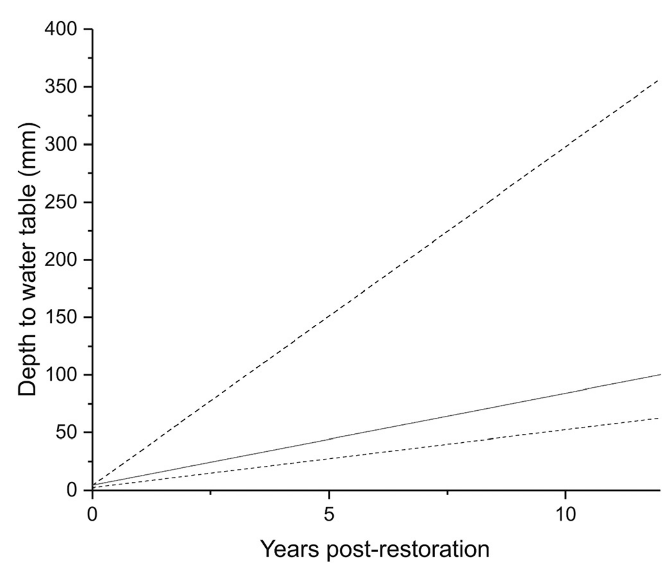

Fig. 4. Normalised median manual depth to water table trajectory at restoration sites relative to control, across all sites with the interquartile range defining error. Higher values on the Y axis reflect reductions in depth to water table (i.e. higher water table) at the treatment site or increases (i.e. lower water table) at the control site. Linear extrapolation was used to produce a median trajectory for all sites.

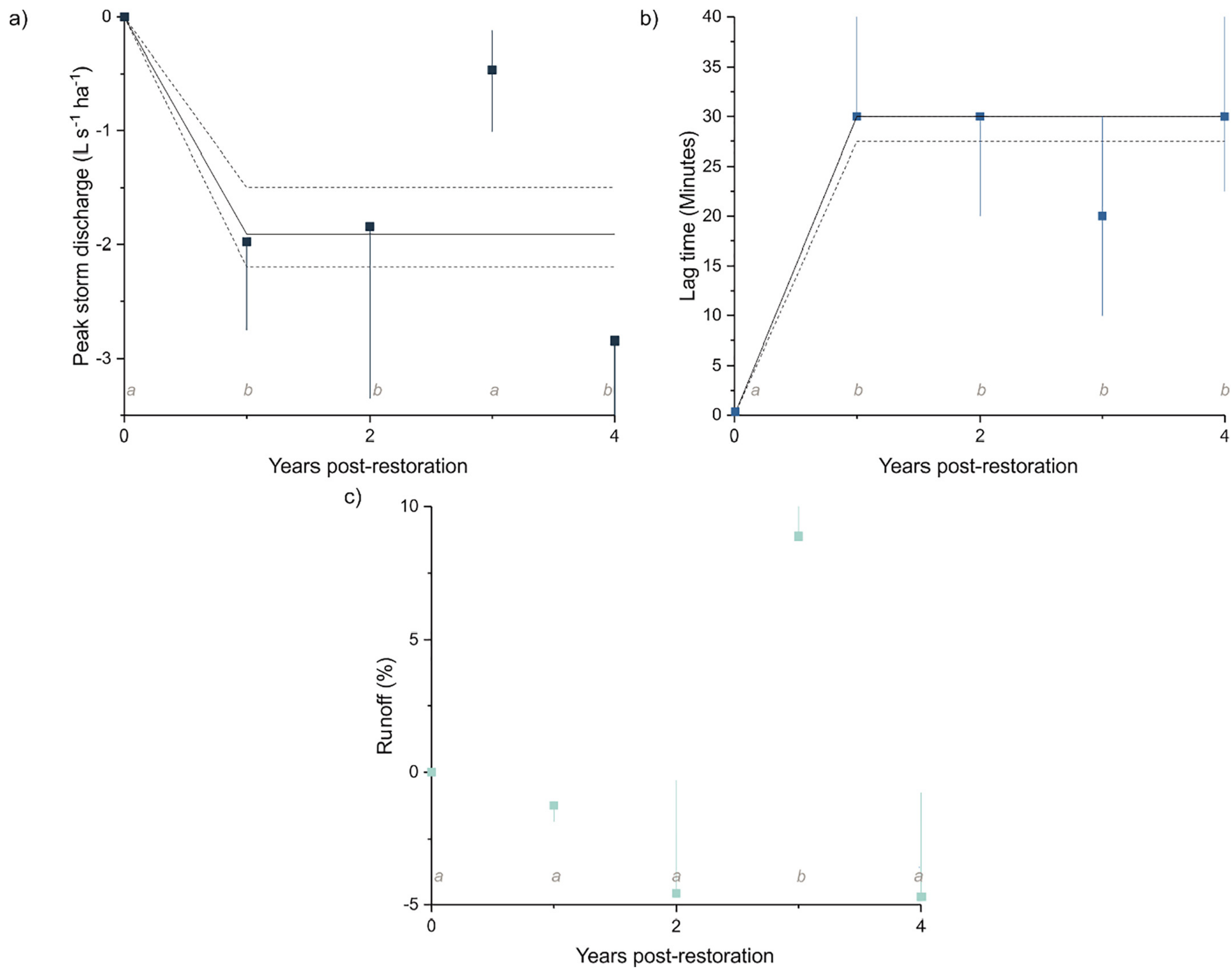

Fig. 5. Runoff characteristic trajectories over four years post-restoration at Kinder Scout a) peak storm discharge, b) lag time, c) runoff coefficient %. Kruskal Wallis groupings represented by grey letters at the bottom of each sub-figure a) similar to pre-restoration value; b) different to pre-restoration value.

Group b, but one of the post-restoration years (year 3) is grouped with the pre-restoration data (Group a). Interestingly, this anomalous post-restoration year is consistent across all metrics, producing the smallest relative difference in lag time, and significantly different runoff coefficient %, and corresponds to an anomalously wet year in the UK record (2014 was the fourth wettest year in the UK series from 1910 - Met Office, 2014). The simplest explanation of the observed data is a step change in lag time and peak discharge in response to restoration, with subsequent variability interpreted as inter-annual noise, resulting from variation in the number and style of storms available for analysis.

Thus, the step change in storm runoff following restoration can be characterised by an increase in lag time of 30 min and a decrease in peak storm discharge of by 1.91 L s^{-1} ha^{-1}, relative to control.

3.4. Fluvial carbon flux

POC data from the sites were relatively limited consisting of two surveys, 2 (Kinder Scout) and 9 (Bleaklow) years post-restoration. No pre-restoration data existed for POC concentrations, therefore the difference between an eroding pre-restoration site and a bare peat control were assumed to be zero, with median POC concentrations from 2 and 9 years post-restoration plotted relative to this (Fig. 6a). An independent *t*-test demonstrated there was no significant difference between year 2 and year 9 ($p = 0.767$), and as such a step change after 2 years restoration is inferred.

No statistically significant trend is present for DOC concentrations post-restoration (spearman rank correlation, $p = 0.505$). Similarities/differences between years examined using a Kruskal-Wallis ANOVA also demonstrated there was no significant difference between years ($p = 0.374$).

Trajectories for fluvial organic carbon flux after peatland restoration demonstrate no significant trend for DOC concentrations but are consistent with a step-change occurring in the first two years post-restoration for POC.

3.5. Raw data

Representing treatment data relative to control data is essential to understanding real trends in the data and distinguishing these from environmental noise. However, in order to provide context, Appendix A presents raw post-restoration data from study sites at Bleaklow or Kinder Scout for each metric at the treatment and control sites. Caution

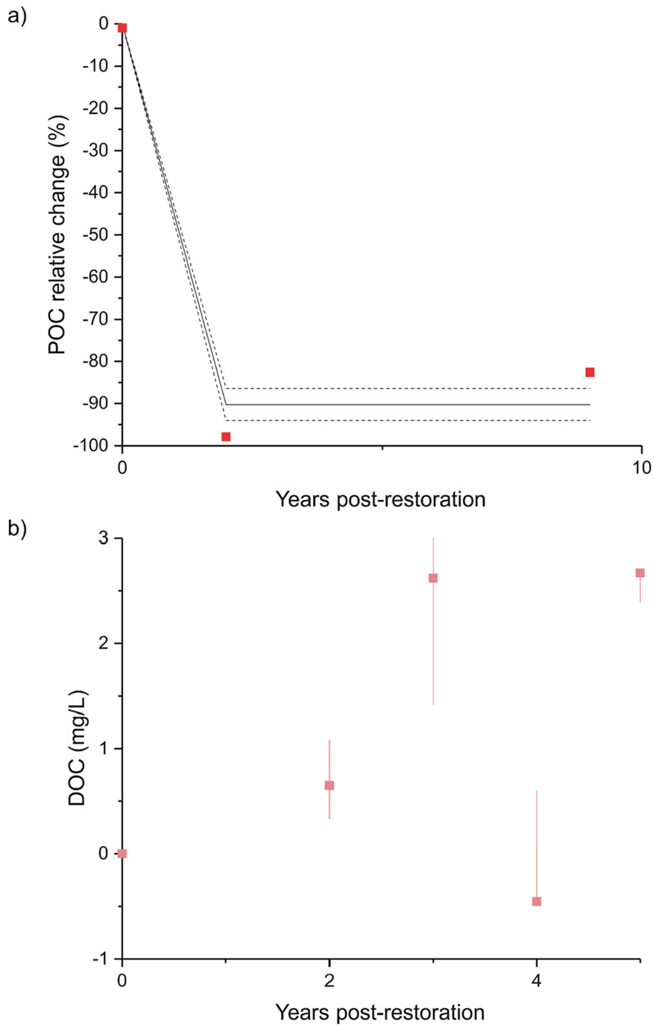

Fig. 6. Fluvial organic carbon flux trajectories normalised relative to control: a) POC % at Kinder Scout (2 years post-restoration) and Bleaklow (9 years post-restoration); b) normalised median DOC concentrations at Kinder Scout.

should be taken in comparing these treatment data to other sites, as they are not relative to control data and absolute values will vary on a site-specific basis.

4. Discussion

4.1. Patterns of ecosystem change from composite datasets

The data presented in this paper synthesises the best available current data to develop understanding of ecosystem responses of vegetation coverage and indicator species; water table level, runoff patterns and water quality alteration, to the LSFM treatment in a severely eroded peatland system. The reactions of the various metrics to restoration vary in their behaviour, both with regards to the timing of first response and the rate of change of the subsequent trajectory towards a new state. Bringing together data from diverse datasets has allowed identification of characteristic timescales of change in response to restoration across a range of ecosystem parameters.

Three parameters respond very rapidly to restoration; percentage vegetation cover (Fig. 3), runoff (Fig. 5) and sediment yield (Fig. 6a) showing large changes within two years of the application of the LSFM restoration approach to bare peat sites. While near complete vegetation cover is achieved relatively quickly (6 years; Fig. 3a), transformation of vegetation cover as indicated by percentage cover of indicator species and indicator species richness occurs more slowly over timescales of up to 12 years (Fig. 3b and c). In contrast, changes in water table (Fig. 4) are slight, but apparently still ongoing 12 years after restoration and changes in DOC flux from the system are not significant five years after restoration (Fig. 6b).

The analysis of the trajectories of ecosystem recovery in this work use a compilation of datasets acquired from a variety of funded projects, carried out by a mixture of practitioners and academics. Ideally, data would be derived from a single project with similar objectives and study design, carried out consistently over a long-term timescale. In real-world practical restoration projects, this is not pragmatic, or often even attainable. Approaches to the collation and synthesis of data which is of high quality, but variable in terms of spatial and temporal coverage, are therefore required. Long term trends can be difficult to discern in such 'noisy' and varied datasets given the natural variability of hydrological and biochemical functions of peatlands, driven by temporal changes in local hydroclimatic conditions. We have accounted for this by considering changes relative to untreated control sites. Indeed, the availability of control data has been key to deriving such clear trajectories of change and should be considered essential in the monitoring and reporting of the impacts of ecological restoration. Continued monitoring over extended timescales than within this study would provide greater confidence in some of the less well constrained trajectories.

Despite the variable number of sites and number of measurements within and between metrics in this study, the vegetation and depth to water table datasets provided consistent directional trends (Figs. 3 and 4). Vegetation cover and water table are commonly regarded as the principal biophysical determinants of peatland function (Holden et al., 2011), so the clear patterns established for a broad geographical coverage of sites for these parameters give confidence in the wider representativeness of these results. For other parameters (e.g. runoff and DOC flux; Figs. 5 and 6b), confidence in the trends established from more local data is derived from understanding potential linkages between these patterns and the key biophysical drivers. The trajectory data resulting from these analyses also offers the potential to highlight the nature of these linkages and this is explored in more detail in Section 4.2.

While there are challenges in the synthesis of diverse datasets, practical restoration work offers the opportunity to undertake controlled manipulations of the landscape at a larger scale than is typically possible in pure scientific work (Dickens and Suding, 2013). Well-designed restoration research, such as that compiled in this paper, offers the potential not only to inform restoration targets, but also to develop scientific understanding of ecosystem processes and their response to restoration actions.

4.2. Inferring process from trajectories of change

Fundamentally the LSFM approach aims to establish vegetation cover (nurse crop) on bare peat, and to create conditions for natural succession, to drive a transition to a more natural moorland vegetation type (Anderson et al., 2009). What is clear from the data presented in this paper, is that re-vegetation of bare peat is a relatively rapid and non-linear process. Close to 100% vegetation cover (expressed as a reduction in bare peat surface) is established within 6 years of initial treatment, with most rapid expansion of cover in the first two years (Fig. 3a). Succession to a more natural moorland vegetation, with significant presence of key indicator species, follows a dose-response trajectory with a non-linear increase in species richness with an asymptote around 12 years. By this time (c. 9 years), the percentage cover of indicator species has approached 100% or above, as a result of overlapping canopies.

The dose response trend is a common measure of vegetation establishment, consisting of a gradually accelerating vegetation coverage response to improving conditions, followed by a stabilisation as competition is introduced (Clewell and Aronson, 2013). The mismatch between non-linear increases in recruitment of new species, and a linear increase in indicator cover, suggests that the nurse crop cover is effectively creating conditions for establishment (rapid recruitment of species), but that it is also potentially competitively slowing the rate of spread of the incoming species. After 9 years vegetation cover has transitioned from bare peat to an apparently stable community of moorland indicator species (Fig. 3b and c). The LSFM restoration model is built around the concept that establishment of a 'nurse crop' allows recruitment of native species, which will eventually out-compete the nurse crop in nutrient poor peatland conditions (Buckler et al., 2008). The data presented here provide some empirical support for this method.

Apparent from the sequencing of trajectories is that rapid changes in POC flux, and the attenuation of storm flow (represented here by an increase in lag time and reduction in peak storm discharge) occur as a step-change early in the phase of vegetation establishment, rather than as a response to the formation of stable moorland communities. POC flux falls by an order of magnitude within the first two years of re-vegetation (Fig. 6). Similarly, there is also an increase in the lag time and decrease in peak storm discharge recorded in flood hydrographs at the hillslope scale (Fig. 5).

These processes are almost certainly linked to an increase in surface roughness from early colonisation of non-moorland species, which has the effects of both slowing the flow of surface generated runoff, and creating conditions where absolute flow velocity is reduced, and suspended POC is deposited. The role of surface roughness in mediating flow velocity and sediment transport has been observed in soil erosion and hillslope runoff studies (e.g. Gomi et al., 2008; Smets et al., 2008; Gumiere et al., 2011; Pan et al., 2016), and these effects have also been observed in natural and experimental peatland systems (Holden et al., 2008; Grayson et al., 2010). Further erosion is also alleviated by the protection provided by vegetation cover and the binding of the soil by roots (Shuttleworth et al., 2017).

Blanket peatlands are known to be naturally hydrologically 'flashy' systems (Evans et al., 1999; Holden and Burt, 2003), but it is clear that restoration has reduced peak flow and increased lag times, overall attenuating storm hydrographs. The relationship between re-vegetation and runoff at these sites is explored in more detail in Shuttleworth et al. (2019).

Water table trajectories display a clear linear change. We might expect a progressive decline in the rate of change over time in water table recovery, mirroring the trend seen in the vegetation data. However, the data derived from well-established restoration sites do not support this. Water table data measured in years 8 to 12 still demonstrates rising water tables, of a comparable magnitude to sites at early stages of post-restoration recovery. There is considerable variability in hydrological recovery between sites (exemplified by the diverging confidence intervals in Fig. 4), but the overall trend across diverse locations with multiple control sites, is that post-restoration there is a small but ongoing positive change in water table relative to control sites.

Water tables close to the surface are consistent with intact peatlands, and therefore recovering ecosystems in peatlands are typically associated with recovering water tables (Price et al., 2016). However, the changes in water table observed in the Peak District are out of phase with changes in vegetation cover, with water table continuing to rise after both vegetation cover and indicator species cover stabilise. If water table changes were driven by changes in energy balance or evapotranspiration associated with changing vegetation cover, we might expect a progressive decline in the rate of change over time as the nurse crop establishes, followed by the native species colonisation. This is not indicated by our data, as small but continuous increases are observed over 12 years (Fig. 4).

These water table results are somewhat counter-intuitive. The slower rate of change than other parameters suggests that the impact of re-vegetation on water table is mediated through sub surface rather than surface effects. At the surface, the change from a smooth brown peat to a rough green sward is rapid, but the impact of re-vegetation on soil structure and hydrological function will develop more slowly as fresh organic matter accumulates, root systems develop and peat re-wets. It is therefore hypothesised that the water table changes observed here are related to changes in soil hydrological function, evolving over time. Full hydrological recovery may not be possible, as hydraulic damage has been shown to not necessarily be entirely reversible when rewetting (Ketcheson and Price, 2011). Monitoring at the timescales constrained by this project is clearly insufficient to assess the drivers behind the water table recovery, and more focussed process based work is required.

There is no statistically significant pattern of change in DOC concentrations over five years. This is surprising as the addition of significant labile carbon to the system through re-vegetation might be expected to increase DOC directly, or through the priming effect (Kuzyakov et al., 2000). Previous work has also shown that re-wetting of previously drained peat can produce spikes in DOC concentration (Worrall et al., 2007). In contrast, the slowly rising water tables discussed previously might be expected to reduce DOC concentrations, by minimising the proportion of the peat mass subject to aerobic decomposition, a standard concept of the diplotelmic model (Ingram, 1978). It should be noted that despite there being no statistically significant change, DOC at the sites studied was slightly elevated relative to control five years after restoration. Relatively slow response of DOC concentrations to re-vegetation might be expected in the same manner as hypothesised for the water table; i.e. changes in DOC production are likely to be a function of slow change in sub-surface conditions, rather than rapid change observed at the peatland surface.

Comparison of the empirically derived trajectories in this study offers a unique way of developing process insight from the data. Fig. 7 presents a conceptual diagram of the response of ecosystem processes to the LSFM restoration approach considered in this study. The focus in this conceptualisation is on the form and the phasing of the ecosystem responses to distinguish transitory from equilibrium effects. The extremes of this model represent a 'restored' ecosystem, while the origin represents a 'degraded' ecosystem (i.e. where no significant improvements of peatland function have been observed).

The patterns of ecosystem change identified in this conceptual model suggest three key timescales define the response of the peatland system to LSFM restoration: 1) at timescales of circa 2 years rapid change in surface conditions through establishment of nurse crop cover; 2) at timescales of circa a decade re-establishment of indicator species richness to levels close to intact peatland sites; and 3) at timescales in excess of a decade ongoing recovery of subsurface hydrological function causes water tables to continue to rise at these timescales.

4.3. Restoration targets

Setting appropriate, clear, and achievable restoration targets is a key component of any practical restoration programme (Hobbs and Harris, 2001). The dynamic nature of ecosystems means that there is not a 'one size fits all method' in ecological restoration. It is important to be realistic when designing restoration programmes to select desirable outcomes achievable in the future, and these may not be entirely based on past environmental status (Thorpe and Stanley, 2011), which may be unattainable (Harris et al., 2006). In some circumstances defining the desired state of restoration in relation to intact systems is impossible, particularly where the desired state is pre-human intervention (Hobbs, 2007). In the instance of the degraded peatlands analysed in this study, the failure to achieve significant change in DOC

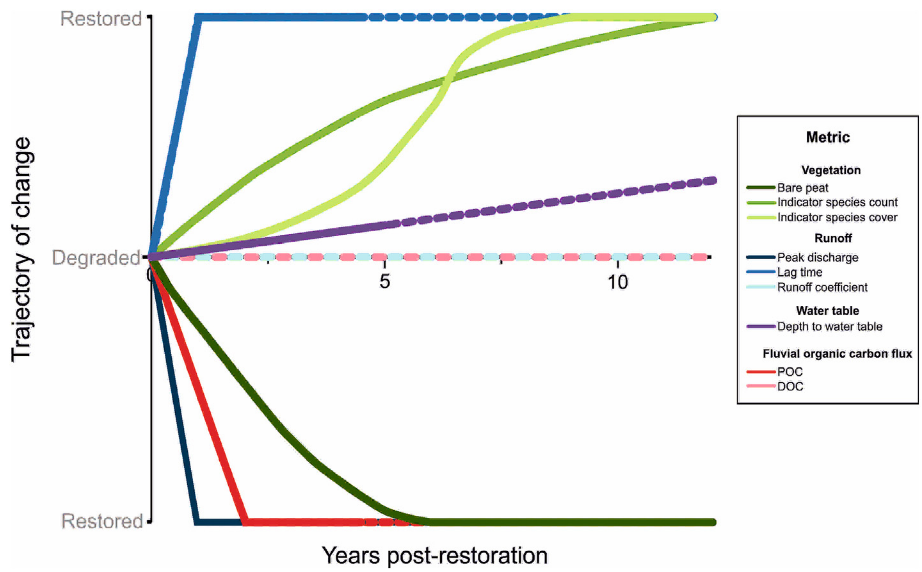

Fig. 7. Conceptual model of trajectory of change of key metrics representing ecosystem services in response to restoration. The extremes of the y axis represent a restored ecosystem whereas the origin characterises a degraded ecosystem. DOC and Runoff % coefficient lie at the origin as there are not significant trends established from the data. Runoff trajectories (from four years onwards), water quality trajectories (from five years onwards) and depth to water table (from five years onwards) are displayed as dotted lines to represent forecasted trajectories.

concentrations over the study period is perhaps unsurprising given the extensive modification of topography and drainage that extensive gully erosion of peatlands produces (Evans and Warburton, 2007). Defining achievable trajectories of ecosystem services based on monitoring of pathfinder restoration schemes (such as the work in the south Pennines reported here) represents an alternative approach to determining realistic restoration goals for such systems.

The LSFM restoration approach for eroded peatlands cannot restore pre-erosion function because of the irreversible (at century timescales) modification of topography but it does deliver significant enhancements to the delivery of key ecosystem services. The development of the 'alternative stable states' concept in restoration ecology has allowed the notion of multiple end points for restoration projects (Beisner et al., 2003). LSFM restoration is delivering large scale rehabilitation (Hobbs and Cramer, 2008) of peatland landscapes. Fig. 7 illustrates that at timescales of around a decade these systems achieve a new steady state for most ecosystem functions. Only the depth to water table metric shows longer term change. The data presented suggest that initial indications of restoration success in terms of establishing vegetation cover are apparent at timescales of 2 years but that longer term monitoring will be required to demonstrate new steady states. For water table it seems likely that long term change in the peatland hydrological cycle means that for this parameter, on present evidence, the restoration may be regarded as an open-ended process (Hughes et al., 2011).

Evaluation of restoration outcomes is a necessary output required by funding bodies and interested stakeholders (Aradottir and Hagen, 2013). Funding timeframes are typically 3–5 years in length. Often this timeframe is too short to implement, monitor and report on the results of large-scale restoration projects. Through the analysis of geographically extensive long-term data we have shown how some benchmarks of successful ecological restoration may be met in short-term timeframes defined by funding bodies. Peatland restoration aims typically include enhancement of a range of peatland ecosystem services such as provision of clean water, carbon storage, runoff regulation and biodiversity with key indicator species (Bonn et al., 2016). The data presented here indicate that demonstration of all these outcomes at project timescales is not possible, but we have identified key benchmarks, that may lead to progressive improvements, even after funding timescales have passed (Hughes et al., 2011).

In blanket peatland systems, the benchmarks that are attainable within the timeframe and scope of short-term restoration programmes are initial revegetation of eroding surfaces, runoff attenuation, and reduction in POC flux. Major changes are observable in these parameters within two years post-restoration (Fig. 7). Understanding of process controls of these changes, through the phasing of the trajectories, points to the establishment of vegetation cover, and consequent changes in surface roughness as a key control. Therefore, within short project timescales, the primary focus of monitoring for peatland restoration using the LSFM approach should be measurement of vegetation establishment. This is in contrast to the general understanding that the essential characteristics of functioning peatland systems, are defined by a high and functioning water table (Holden et al., 2011), and the presence of keystone peatland species (Evans and Warburton, 2007).

Over short timescales the trajectories established in this study indicate that measurement of water table and biodiversity, while desirable to produce baseline data, could lead to false negatives in terms of confirming restoration success. Timescales for significant responses of these variables are in the order of as little as 2 years for initial change and approximately 9 years for the levelling off of trajectories such as biodiversity goals. Arguably the key to the provision of a number of the ecosystem services in this environment is vegetation cover, but not necessarily native vegetation cover. It is sufficient in these environments to begin the process of re-vegetation with non-native species, and the trajectories presented here indicate that the system will evolve towards a more characteristic peatland system over time.

5. Conclusion

Peatlands have been the priority of a number of high-profile restoration projects in the UK, due to the nature and severity of degradation of the landscape, and the wealth of ecosystem services that these systems provide. This paper, for the first time, uses comprehensive data from such projects to report on the impact of the widely adopted LSFM approach of restoring bare peat surfaces. Despite the diversity of data sources, consistent comparison to control data has allowed this study to rigorously assess ecosystem trajectories in response to LSFM treatment. Practitioners and scientists have a wealth of data that can be utilised to define restoration targets. This study has shown that combining large and consistent datasets amplifies the signal in noisy data, to identify dominant trends in the delivery of key ecosystem services.

These trends demonstrate that the establishment of vegetation cover and the consequential development of surface roughness are key drivers in rapidly reducing particulate carbon loss and attenuating stormflow. Over time, biodiversity is improved by the return of native species, and hydrological function gradually recovers. While the formation of new equilibrium conditions exceed a decade, observation of established vegetation cover over timescales of up to two years is sufficient for reporting on project success, for typical funding timeframes.

The trajectories developed in this study are important both to help guide practitioners of ecological restoration, and to set future research agendas for peatland scientists. Phasing of ecosystem service responses indicates that there are different characteristic timescales for the recovery of peatland functions driven by surface and subsurface processes. The results presented in this paper support the wider scale adoption of the LSFM approach on eroding blanket bog peatlands globally. These data support the key role for surface re-vegetation as an important control on peatland ecosystems services and identify an important research agenda to understand processes and rates of change in the subsurface soil system in response to peatland restoration.

CRediT authorship contribution statement

Danielle M. Alderson: Methodology, Formal analysis, Data curation, Writing - original draft, Writing - review & editing, Visualization. **Martin G. Evans:** Conceptualization, Methodology, Formal analysis, Investigation, Data curation, Writing - original draft, Writing - review & editing, Visualization, Supervision, Project administration, Funding acquisition. **Emma L. Shuttleworth:** Conceptualization, Methodology, Formal analysis, Investigation, Data curation, Writing - original draft, Writing - review & editing, Visualization, Supervision, Project administration, Funding acquisition. **Michael Pilkington:** Conceptualization, Methodology, Formal analysis, Investigation, Data curation, Writing - review & editing, Visualization, Supervision, Project administration, Funding acquisition.
Tom Spencer: Methodology, Formal analysis, Investigation, Data curation. **Jonathan Walker:** Conceptualization, Methodology, Supervision, Project administration, Funding acquisition. **Timothy E.H. Allott:** Conceptualization, Methodology, Formal analysis, Investigation, Data curation, Writing - review & editing, Visualization, Supervision, Project administration, Funding acquisition.

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Appendix A

Summary statistics for raw metrics at control and treatment sites for the last available year of post-restoration data (i.e. the end point of the trajectories presented in the paper). Please note, the trajectories were derived from multiple sites which represent varying stages of restoration, but these 'end member' data were available only for single sites. These data are provided to allow comparison between re-vegetated and bare sites, but caution must be exercised when interpreting the differences as pre-restoration data are not available for many of these late-stage restoration sites.

Metric	Years since restoration	Site	Control			Treatment		
			Post-restoration value	Lower Quartile	Upper Quartile	Post-restoration value	Lower Quartile	Upper Quartile
Bare peat cover (%)[a]	11	Bleaklow	98.5	97.6	99.4	0	0	2
Indicator species count[a]	11	Bleaklow	0.00	0.00	0.75	4	4	5
Indicator species cover (%)[a]	11	Bleaklow	0.00	0.00	0.38	109	47.5	168
Depth to water table: early-stage (mm)[b]	5	Kinder Scout	267	251	346	200	184	245
Depth to water table: late-stage (mm)[b]	12	Bleaklow	416	376	463	289	268	306
Peak storm discharge (L s^{-1} ha^{-1})[c]	4	Kinder Scout	3.71	2.17	5.52	1.51	1.01	3.26
Lag time (minutes)[c]	4	Kinder Scout	35	15	138	55	35	165
Runoff (%)[c]	4	Kinder Scout	29.4	21.2	37.0	35.2	24.3	39.1
DOC (mg/L)[d]	5	Kinder Scout	24	23	25	27	26	29
POC (g per week)[e]	9	Bleaklow	0.37	0.25	0.57	0.05	0.03	0.07

[a] Derived from quadrats during yearly campaigns. Treatment ($n = 41$); control ($n = 10$).
[b] Derived from autumn manual dipwell campaigns. Early-stage treatment ($n = 7$) and control ($n = 7$); late-stage treatment ($n = 12$) and control ($n = 12$).
[c] Derived from paired storms from data collected throughout a complete year. Treatment ($n = 30$); control ($n = 30$).
[d] Derived from monthly sampling campaign. Treatment ($n = 5$); control ($n = 5$).
[e] Derived from multiple sediment traps over 5 sampling campaigns ($n = 5$).

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