Meta-analysis reveals that enhanced practices accelerate vegetation recovery during peatland restoration

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Abstract

The provision of critical ecosystem services like carbon sequestration by peatlands has been degraded around the globe. Peatland restoration represents an opportunity to tackle the twin global emergencies of climate breakdown and biodiversity decline. Nonetheless, restoration success relies on a sound understanding of recovery trajectories associated with different restoration techniques. Focussing on temperate/boreal *Sphagnum*-dominated peatlands, we used a quantitative meta-analysis of 28 studies representing 275 sites in 11 countries to test for effects of peatland status (intact, restored, degraded), varying restoration interventions and time since restoration on vegetation as a key indicator of peatland condition and functioning. Enhanced restoration (such as active revegetation) resulted in recovery to pre-disturbance levels within 30-35 years for *Sphagnum* mosses, and 20-25 years for many other peatland specialist species, and was the only restoration approach where positive outcomes were seen across all vegetation response variables. The use of standard restoration techniques, such as rewetting, was projected to result in cover of *Sphagnum* mosses and peatland specialist plants reaching that of intact sites within 45-55 years post-restoration. Passive restoration (cessation of the degrading activity with no active restoration) generally elicited limited recovery of keystone peatland vegetation (*Sphagnum* spp.) even after multiple decades. A lack of standardisation in monitoring severely constrains the analysis of peatland restoration outcomes. Increased funding for monitoring and reporting outcomes, and improved monitoring consistency, could greatly enhance our understandings of peatland restoration ecology and improve practice.

Key words: biodiversity, climate change, fen, mire, rewetting, *Sphagnum*

Implications for practice

- Active reintroduction of peatland plants such as *Sphagnum* mosses successfully accelerates the re-establishment of peatland vegetation cover.
- It remains uncertain whether, and over what timescales, passive restoration enables a peatland to recover.
- While long-term data for post-restoration vegetation recovery remains limited, the sharing of such data that does exist is urgently needed – as are strengthened connections between restoration researchers and practitioners.
- Increased funding for monitoring and reporting restoration outcomes, and standardisation of monitoring, would enable improved integration of data.

Introduction

Peatlands cover <3% of the global land area (Xu et al. 2018), where they provide crucial ecosystem services (Bonn et al. 2016; UNEP 2022). These habitats contain a globally significant carbon stock (Yu et al. 2012), harbouring the largest C density of any terrestrial ecosystem (Joosten et al. 2016). Additionally, peatlands contribute to sustainable water provision (Parry et al. 2014; Wilson et al. 2011) and flood regulation (Wilson et al. 2011), and support a highly specialised flora and fauna (Rydin & Jeglum 2013; Minayeva et al. 2017). These vital habitats are vulnerable to perturbation (Parry et al. 2014) and highly threatened (Reed et al. 2014).

Peatlands have been degraded by direct and indirect human activities including drainage and conversion for agriculture, forestry and mining (Anderson & Peace 2017; Chimner et al. 2017), extraction for horticulture, animal bedding and fuel (Cruickshank et al. 1995; Chapman et al. 2003), fire (Glaves et al. 2013; Turetsky et al. 2014; Douglas et al., 2015), by nutrient enrichment (McBride et al. 2011), changing climatic conditions (Heijmans et al. 2008) and general atmospheric pollution (Smart et al. 2010). It is estimated that 12% of global peatland has been degraded (UNEP 2022).

Humanity faces two interlinked crises – climate breakdown and biodiversity loss – with catastrophic implications (IPCC 2021; IPBES 2019). Solutions that could help to mitigate both these threats are urgently needed (Soto-Navarro et al. 2020; WWF 2020). Restoring degraded peatlands represents such a nature-based solution to addressing these crises that also potentially enhances the regulation of pests and diseases (Gilbert 2013). These benefits have strongly increased the profile of peatland restoration (Bullock et al. 2012; Rochefort & Andersen 2017) and the political (e.g. Defra 2021; European Commission 2021) and research interest in doing so

(Andersen et al. 2017). The UN Decade on Ecosystem Restoration 2021-2030 is a rallying call for the revival of ecosystems, and the drive for peatland restoration is expected to persist over the next half-century (Grzybowski & Glińska-Lewczuk 2020). Nonetheless, the underpinning evidence-base to inform peatland restoration has been relatively limited (Taylor et al 2018), with activities regularly relying on trial and error (Lamers et al. 2015). Collective evidence and robust monitoring frameworks are urgently needed to improve the effectiveness of future interventions (Salafsky et al. 2019; UNEP 2022).

Peatlands are particularly prevalent in cool and wet regions in the Northern Hemisphere (Xu et al. 2018; Holden 2005; Joosten 2008). We therefore focussed our study on the Northern Hemisphere's temperate and boreal peatlands, comprising bogs and fens with peat-forming *Sphagnum* mosses as keystone species and ecosystem engineers (Van Breemen 1995; Rochefort 2000; Caporn et al. 2018), and with other temperate and boreal peatland specialist plants. The successful restoration of peatlands and associated ecosystem services requires the recovery of characteristic, self-regulatory peat-forming vegetation (Rochefort 2000; Littlewood et al. 2010).

Range and intensity of peatland restoration techniques vary. They are often tailored to the specific type of degradation (Bonn et al. 2016). Some extracted peatlands may simply be abandoned (Poulin et al. 2005) with the expectation that peatland species re-establish spontaneously ('passively') from remnant vegetation where conditions are favourable (Lavoie et al. 2003, Minayeva et al. 2017). This approach requires little resources, but reduces predictability (Graf et al. 2008) and may be insufficient for reversing degradation over current monitoring timescales of several decades. It is therefore crucial to identify specific drivers that support the regeneration of characteristic *Sphagnum* carpets and the return of typical peatland vascular plant assemblages. Rewetting is common and underpins most restoration efforts (Lunt et al. 2010; Taylor et al. 2019); drainage ditches are commonly blocked using peat, wood, or heather bales to raise the water table (Price et al. 2003; Armstrong et al. 2009). Afforestation of peatlands causes drastic abiotic and biotic changes through drainage, ploughing, and subsequent tree planting of often non-native, commercial species (Hancock et al. 2018; Anderson & Peace 2017). To restore afforested peatlands to their original conditions, tree felling is likely required, especially as some tree species can tolerate the waterlogged soils generated by rewetting (Anderson & Peace 2017). Restoration may also include active revegetation, including reintroduction of target species such as *Sphagnum* mosses (Rochefort et al. 2003; Rochefort & Lode 2006).

To ensure the effective deployment of future restoration and the investment underpinning it, it is crucial that restoration outcomes are understood (Parry et al. 2014; Rochefort & Andersen 2017). Vegetation change, the focus of this paper, is often monitored as a principal determinant of peatland functionality, including of carbon sequestration (Holden et al. 2011; Swenson et al. 2019), with the regeneration of *Sphagnum* coverage a key indicator of potential for peat formation (Rochefort 2000; Poulin et al. 2012; Lindsay et al. 2014). *Sphagnum* mosses are sensitive to water-table changes (Rydin & Jeglum 2013), thus also acting as proxies for hydrodynamics. Evidence of general vegetation responses to peatland restoration has been collated in previous reviews (e.g. Taylor et al. 2019; Rowland et al. 2021; Kreyling et al. 2021). However, more detailed, quantitative syntheses of specialist peatland plants including *Sphagnum*, restoration trajectories and formal testing of differences between restoration techniques across the temperate Holarctic region, remain scarce. We therefore extend previous reviews using a quantitative meta-analysis of vegetation responses to peatland restoration.

Using metrics of peatland vegetation cover and plant species richness, we address the following questions: (1) how does vegetation differ between degraded, restored and intact peatlands; (2) how long does it take for the vegetation of damaged peatlands undergoing restoration to resemble that of comparable intact ones; and (3) how does this timeframe differ between restoration techniques of different intensities? We predict that more intensive restoration techniques, including active revegetation, deliver faster recovery of characteristic peatland plant communities.

Methods

A systematic literature search was conducted in Web of Science, seeking primary studies published in English from 1981 to mid-2021 that describe peatland restoration and vegetation (see Table S1 for terms). The initial search, undertaken for a wider review of biodiversity responses to peatland restoration (Douglas et al. 2019), included plant and non-plant taxa; non-plant taxa were excluded from the current review. Papers were retained using a hierarchical approach; retained if title suggested fit to scope; if title insufficient to decide, abstract read; if title and abstract insufficient, paper skimmed. This yielded 272 papers which were filtered to include papers relating to temperate peatlands; involved restoration of degraded peatlands; included a vegetation response; presented primary findings. This resulted in 142 retained papers.

The following information was extracted per study: status of study sites (intact, restored, degraded); restoration technique/s; monitoring age (i.e. years since intervention commenced or degradation ceased). If data was not provided within a paper, or its supporting information, the study author was contacted to seek it. In some cases, unpublished data obtained after the paper was published were included if it fitted the required characteristics. Sometimes a range of monitoring age was reported; to make best use of data, the mid-point was used. Latitude of study sites was extracted where provided or calculated based on description of study location, because this could affect vegetation growth rates and hence recovery trajectories (Xu et al. 2017). We limited the latitudinal range of studies to between 45ºN and 65ºN, to target Holarctic peatlands (Figure 1). The papers included in the meta-analysis are summarised in Table 1, with further detail in Table S2. This resulted in the inclusion of 275 restoration sites, with a total of 5929 monitoring plots within those sites.

To make best use of resulting sample sizes we focussed on four response variables of percentage cover and diversity (species richness) of both *Sphagnum* mosses and a wider suite of characteristic Holarctic peatland species (including *Sphagnum* spp.), allowing us to use 28 studies. *Sphagnum* cover was frequently reported at genus rather than species level, making it impossible to differentiate between *Sphagnum* species with different traits. Nonetheless, as a group, their overall cover still represents an accepted indicator of peatland status (Rochefort 2000; Poulin et al. 2012). Species assemblages on intact peatlands are often distinct but species-poor (Minayeva et al. 2017; Strobl et al. 2019). For this reason, total species richness is considered a poor indicator of condition as it could include specialists and generalists, the latter sometimes adapted to degraded conditions and showing differential responses to restoration (e.g. Ilmonen et al 2013). We therefore consider our measure of richness of characteristic peatland specialists appropriate. The full list of plant species was reviewed and ‘peatland specialists’ (e.g. *Sphagnum* spp., *Drosera* spp.) were identified with reference to the source studies, and where necessary other available literature (e.g. British Bryological Society 2021). The list of species and our categorisation are presented in Table S3. Only species-level data was included, as genera regularly include both peatland and non-peatland species.

Analyses

The four vegetation response variables were calculated per plot as appropriate (Table S4). We used Generalised Linear Mixed Models (GLMM) using the 'lme4' package in R (Version 4.0.3) (R Core Team 2020) (script in Figure S1), testing for effects relating to peatland status (intact, restoration, degraded), time since restoration and between different restoration interventions (Table 2). For the first analysis (Table 2a) any sites reported as 'undisturbed' were classed as 'intact'. Of the rest, any sites with a reported 'monitoring age' were classed as a 'restoration site' (so encompassing the full range of restoration approaches), and the rest as 'degraded' (i.e. assumed that no management actions had been undertaken to facilitate recovery). The latter two sets of analyses (Table 2b and c) compared restoration against degraded sites, excluding intact sites as these do not have useful monitoring ages; instead, data regarding intact sites was used to calculate a reference level. Site ID was a random effect in all tests to account for variability between studies and local conditions. Latitude, as a main effect and interacting with monitoring age, showed no significant association with any response variables and was excluded from further modelling. Where the three-level peatland status factor (Table 2a) indicated a statistically significant difference between status types, we used robust non-parametric resampling as post-hoc testing (Douglas et al. 2009) to quantify differences between factor levels (Figure S1), calculating the mean fitted response and 95% confidence intervals per factor level.

When testing the effect of different restoration interventions on restoration trajectories (Table 2c), the relatively large number of different interventions were consolidated and classed into three broad categories of intervention intensity (Table 1). The 'passive' category captures any sites that have had no reported intervention to stimulate recovery so includes degraded control sites in addition to sites explicitly reported as abandoned. Any sites with reported intervention to remove stressors (e.g. drainage blocking, rewetting, tree felling) were classed as 'basic' restoration, with any that reported measures to actively reinstate peatland ecosystems, by reintroducing vegetation, were classed as 'enhanced'. Where measures had been combined (e.g.

rewetting and active revegetation) the most intensive measure took precedence in the categorisation.

Results

The collected data comprised 5929 data points; 1622 from ‘intact’ peatland sites, 3616 ‘restoration’, and 691 from ‘degraded’ sites. The dataset contained a high proportion of records relating to *Sphagnum* mosses (available for 98.3% of plots), compared with wider species-level data (80.9%). The geographic distribution of study sites was dominated by North America, Western Europe, and Scandinavia (Figure 1), reflecting key elements of the distribution of Holarctic peatlands, but also many of the main locations where studies on peatland degradation and restoration have been conducted.

Effect of peatland status

All studied variables were lowest in degraded peatlands, followed by restored peatlands, and highest in intact peatlands (Figure 2; Table 2a). The magnitude of difference between degraded and restored peatlands was less pronounced for vegetation cover values, while restored and intact peatlands were more similar in terms of richness values, but still significantly different (Figure 2).

The model-fitted cover of Sphagnum mosses in restored peatlands (mean ± SE of 35 ± 0.22) was around a third of that in intact peatlands (118 ± 0.64), and cover of peatland specialist species in restored peatlands (50 ± 0.25) was around half of that in intact peatlands (104 ± 0.54). Restored peatlands included sites of greatly varied restoration age, ranging from 1 to 63 years post-restoration, partly explaining the high variability in responses here.

Post-restoration timescales of vegetation change

Change in all response variables with time since restoration differed significantly between the three levels of restoration intensity we differentiated, and differences between passively and basically restored sites were weaker for measures of species richness than vegetation cover (Table 2c). Only enhanced restoration practices consistently accelerated the recovery of restored peatlands, both in terms of peatland vegetation cover and richness.

Sphagnum cover increased at markedly different rates between treatments. Active revegetation using the ‘Moss Layer Transfer Technique’ (MLTT) greatly accelerated the recovery of the *Sphagnum* cover, with a total moss cover approaching, and potentially even exceeding that found at intact sites, within 35 years (Figure 3a). Removal of stressors with basic levels of restoration intensity (e.g. rewetting, tree felling) also resulted in marked recovery, although models assuming linear recovery trajectories still projected ~50+ years before these sites’ *Sphagnum* cover resembled that of intact sites. Passive restoration was associated with significantly slower recovery, projected in our modelling to occur over centuries rather than decades (Figure 3a).

Under enhanced restoration, *Sphagnum* moss diversity reliably first matched, and then even exceeded levels at corresponding intact peatlands, within a decade (Figure 3c). *Sphagnum* moss diversity appeared to increase marginally quicker following passive restoration than following basic restoration, although with a low degree of statistical confidence (Figure 3c).

The cover of specialist peatland plants was projected to closely resemble that of intact sites ~23 years after enhanced restoration and ~32 years following stressor removal (basic restoration). The modelled trajectory on sites with passive restoration suggests stagnating, and even further deteriorating conditions in this broader category at degraded sites even 50+ years following abandonment (Figure 3b).

Enhanced restoration led reliably to intact levels of peatland specialist plant diversity within ~8 years, while stressor removal (basic restoration) appeared to cause a decline in peatland specialist diversity. The mean response of peatland specialist diversity to passive restoration on average was again slightly negative. However, the character of these responses could not be predicted with high confidence (Figure 3d).

Discussion

The restoration of degraded peatlands, as a nature-based solution, offers a significant opportunity to address both climate change and biodiversity decline, but to harness it to greatest effect it is crucial that evidence-led approaches are adopted (Salafsky et al. 2019; UNEP 2022). Our analysis brings together data from multiple studies, offering insights into the effectiveness of different restoration approaches from this combined evidence-base.

More intense restoration interventions deliver faster changes

Our findings suggest that the recovery of peatland vegetation cover towards that of intact sites is typically a long-term process, spanning multiple decades, reemphasising the need to conserve existing pristine peatlands (Loisel & Gallego-Sala 2022). Importantly, our findings also indicate that more intense restoration interventions can accelerate positive outcomes. Overall, the differences between intact and restored sites remained highly significant, suggesting that it is difficult to reinstate a near-pristine peat-forming cover of *Sphagnum* spp. and other peatland plants over the timeframe afforded to most post-restoration monitoring. Nevertheless, our results suggest that, while the cover of peatland specialist plants and *Sphagnum* mosses require 40-45 years to resemble that of intact peatlands following basic restoration, active revegetation techniques can reduce this time to approximately 35 years.

Species richness of peatland plants also responds positively to enhanced restoration, which our model suggests can reach and exceed that of intact peatlands within a decade. By contrast, passive restoration and basic restoration techniques elicit weaker responses, suggesting that more intensive measures are far more successful in creating a range of microhabitats that favour bryophyte diversity. Although, emerging evidence (Boucher, personal communication) suggests there may be a decrease in diversity with time post-restoration, as pioneer and opportunistic bryophytes were replaced during the expansion of the *Sphagnum* carpet composed of late-successional species becoming dominant.

With enhanced techniques, the recovery of some aspects of underlying peatland functioning, such as carbon sequestration, can occur even more rapidly (Nugent et al. 2018 & 2019) and within the normal range or slightly higher for former raised bogs. Hambley et al. (2019) report a blanket bog site switching from a C source to a sink within 16 years following rewetting and active revegetation, albeit with C sequestration occurring at a lower rate than intact sites, while Nugent et al. (2018) report 14 years. Though this gives some cause for optimism, ecosystems in recovery may not be as stable and resilient as those that are fully-recovered, or intact peatlands (Koebsch et al. 2020) which have withstood natural disturbances for millennia (Alexandrov et al. 2020). This may be particularly true of peatlands undergoing only passive or basic interventions, based on the minimal to negative response of peatland specialist species richness identified here, and the established relationship between biodiversity and ecosystem resilience (Naeem & Li 1997; Ives & Carpenter 2007). Therefore, these sites should not be presumed to be permanent C sinks and vegetation monitoring should be continued. Some resilience of restored peatlands to fire has been demonstrated, but this study was limited to one site that was restored using MLTT (Blier-Langdeau et al. 2022).

It is unsurprising, given that functioning peatlands require a high water table, that rewetting is a common restoration technique to reverse the damage from widespread drainage, nor that it is beneficial (Taylor et al. 2019). Rewetting has been considered to ‘jump-start’ the recovery of peatland ecosystem function (Kareksela et al. 2015), yet our results suggest that as a single technique it is generally unlikely to deliver reliable improvements in the short-term and that active revegetation is additionally required. This may be due to a depleted seedbank or lack of nearby diaspores (Smolders et al. 2003; Hedberg et al. 2012) or in the contrary, the presence of other dominating species (Gaffney et al. 2020), or other limiting factors, such as abiotic disturbances like wave erosion. Whatever the underlying cause, our results suggest that solely restoring the water-table, though a necessary step (Klimkowska et al. 2010; Lunt et al. 2010), does not guarantee recovery of peatland ecosystems (Kreyling et al. 2021).

Further, our modelled species richness responses to basic methods like rewetting may prompt concern. Intact peatlands often include a variety of habitats, for example pools, hollows and hummocks, lagg, patches of tree or shrub thickets giving opportunity for a range of plants with different ecological niches (Glaser 1992). Species richness, both within *Sphagnum* spp. and in the peatland specialist category overall, is a combined measure of species differing in their ecological niches, including some that might tolerate or favour slightly drier, or extremely wet conditions (Andrus et al. 1983; Granath et al. 2010; Hájek & Vicherová 2013). Therefore, these species may persist to varying extents at degraded sites. Habitat heterogeneity may not be reliably reinstated through basic measures alone, indicating a risk of relatively homogenous conditions lacking in the distinctive variations in microtopography, ecological niches, and self-regulatory mechanisms that underpin functioning peatlands (see Pouliot et al. 2012).

The ‘shock’ of rapid hydrological change or sudden exposure of formerly tree-shaded areas could lead to severe temporary declines in remnant peatland plant populations in the immediate post-restoration period (Smolders et al. 2003; Poschlod et al. 2007). These factors may also contribute to the slower increase in peatland species cover compared with active revegetation. An adaptive approach, using targeted transfer and introduction of plants to specific areas of restoration sites that align with their respective ecological requirements (e.g. in response to the altered water-table), could ensure faster and better restoration outcomes.

Passive techniques have limited potential to restore peatland form and functions

Understanding the capacity of spontaneous processes to contribute to restoration is fundamental to decision-making (Prach & Hobbs 2008; Chazdon et al. 2021), helping to identify where active measures are required (Girard et al. 2002). Drainage is a common feature of peatland degradation and at sites where this is not actively reversed, hydrological conditions required for peatland species to establish will not be reinstated; instead, this is likely to provide favourable conditions for non-peatland vascular plants with tolerance for drier conditions (Girard et al. 2002; Poulin et al. 2005). Our results concur with prior studies (e.g. Soro et al. 1999; Poulin et al. 2005; Pouliot et al. 2012) that simply abandoning such sites and awaiting passive recovery, without active reversal of the stressor, generally delivers little recovery of characteristic peatland plants over time.

This low capacity for self-repair could be compounded by changes in climatic conditions, atmospheric nutrient depositions, and sea-level rise, which further increase the potential for succession towards altogether different habitat types like heathland or grassland (Girard et al. 2002; González et al. 2014; Guêné-Nanchen et al. 2020). Given the multitude of pressures already inflicted upon peatlands and their drastic potential abiotic and biotic impacts (Jonsson-Ninniss & Middleton 1991; Lavoie & Rochefort 1996; Girard et al. 2002), the likelihood of peatland recovery without intervention is low other than in very specifically favourable conditions (e.g. Poulin et al. 2005).

Implications for peatland restoration policy, funding, and practice

Self-repair has been considered possible where a peatland is already close to a tipping point (Robert et al. 1999; Milner et al. 2021) – given the lower capital costs of ‘do nothing’ approaches, low or no intervention may be tempting. While acknowledging that a linear recovery trajectory is unlikely in reality, assuming a stable modelled rate of change, we predict that in most scenarios it would take centuries for most degraded sites to resemble an undisturbed peatland without active intervention. This is far too long to meaningfully contribute to the present environmental crises that we face and assist in climate change mitigation. Therefore, based on our findings, funders, policy-makers and practitioners should anticipate that active measures will be needed to deliver peatland restoration goals and secure their essential contribution to climate change mitigation.

Globally, peatland restoration is severely under-funded (UNEP 2021). Our analysis confirms the need for more intensive interventions, which will have economic and practical implications. Although markets for nature-based solutions are emerging, with public, private, and blended finance options becoming available to support peatland restoration (Moxey et al. 2021), further research is urgently needed on the required scale of funding. While there is good evidence regarding the cost of basic measures, such as rewetting (Artz et al. 2018), and some on the cost of enhanced methods (Quinty & Rochefort 2003), there is little recent published analysis of the costs. Innovative revegetation methods (e.g. Caporn et al. 2018) could offer increasingly cost-effective options, therefore ongoing trials and knowledge dissemination may be beneficial (e.g. cultivation of donor material in Sphagnum farms; Gaudig et al. 2017, Guêné-Nanchen & St-Hilaire 2022). In addition to financial implications, the sustainable supply of *Sphagnum* mosses and other peatland plants used in active revegetation should also be appraised as there may be insufficient donor sites (Caporn et al. 2018), which should have sufficient plant diversity (Hugron & Rochefort 2018).

The need for long-term monitoring of restored peatlands and adaptation

Monitoring the biological outcomes of restoration projects is often severely constrained by resource limitations and insufficient funding timescales, and this appears particularly pertinent for slow-developing ecosystems like peatlands (Taylor et al. 2019; Alderson et al. 2019; Douglas et al. 2019). We found that the high variability in sampling and reporting regimes further impedes comparability of existing datasets. In the context of increasing interest in, and implementation of, peatland restoration as part of measures to address climate change and biodiversity collapse, we call for the development of standardised, adequately funded long-term biodiversity monitoring schemes and integration of these into restoration programmes (UNEP 2022). Emerging ecosystem services markets or carbon off-setting schemes that could finance peatland management and restoration also need to be underpinned by effective long-term monitoring (Bonn et al. 2014; Brown 2020). Monitoring itself may include use of techniques working on large spatial scales linked to remote sensing (Burdun et al. in preparation), but basic information on both *Sphagnum* moss cover and the composition of the overall vegetation should also be collected.

Where adaptive re-vegetation measures and active on-site relocation of plant species is undertaken in response to restoration-related changes in habitat conditions, the effects should be monitored using robust, ideally Before-After-Control-Impact (BACI) designs (Douglas et al. 2019) as in Rochefort et al. (2013). Changing environmental conditions, particularly warmer and drier climates, may present additional complexities in the recovery of degraded peatlands (Klimkowska et al. 2010; Thorpe & Stanley 2011; Guêné-Nanchen et al. 2020); monitoring activities must continue for the long-term (>30-40 years) as the trends observed to date may not hold true under differing conditions.

Research on the resilience of restored peatlands is urgently needed (Loisel & Gallego-Sala 2022). As referred to above, we are aware, so far, of only one study assessing the resilience of a restored peatland to fire (Blier-Langdeau et al. 2022). Additionally, there seems to be very limited research solely assessing the impact of climate change in peatland or wetland restoration. We also echo the need identified by Kreyling et al. (2021) for improved understanding of the value and resilience of potentially novel ecosystems resulting from restoration. Indeed, peatland restoration has been shown to create beneficial novel habitats for the declining Savannah sparrow in Canada

(Desrochers & Rochefort 2021), and analyses of such additional, unexpected benefits of habitat restoration efforts should be encouraged. In some scenarios, embracing novel ecosystems and the services they provide seems highly valuable, while it remains vital that decision-making is based on sound science and considers the relative provision of services in which functioning peatlands excel – not least, carbon storage.

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Table 1. Intervention types investigated in meta-analysis studies

Intervention level	Intervention technique	Description	Relevant studies
No intervention/ passive restoration	Abandonment / natural revegetation	No effort is made to remove or reverse stressors that have caused peatland degradation.	Girard et al. (2002); González et al. (2013); Graf et al. (2008); Kollmann & Rasmussen (2012); Nishimura & Tsuyuzaki (2014); Pouliot et al. (2012); Soro et al. (1999)
Stressor removal / basic restoration	Rewetting	Reversal of drainage systems to reinstate the water-table, e.g. through ditch or gully blocking.	Anderson & Peace (2017); Bellamy et al. (2012); Bönsel & Sonneck (2011); Glendinning & Hand (2016); Görn & Fischer (2015); Haapalehto et al. (2017); Hancock et al. (2018); Hedberg et al. (2012); Hynninen et al. (2011); Jauhiainen et al. (2002); Klimkowska et al. (2015); Koslov et al. (2016); Maanavilja et al. (2015); Mälson et al. (2010); Punttila et al. (2016); Putkinen et al. (2018); Strobl et al. (2018)
	Tree felling / removal	Felling of trees and scrub, typically conifers, from afforested peatlands, sometimes with removal of felled material.	Anderson & Peace (2017); Haapalehto et al. (2017); Hancock et al. (2018); Hedberg et al. (2012); Jauhiainen et al. (2002); Punttila et al. (2016); Strobl et al. (2018)
	Fen-specific measures	Use of traditional management techniques, e.g. mowing, to reinstate conditions required for characteristic plant species.	Klimkowska et al. (2015); Mälson et al. (2010); Ross et al. (2019)
Enhanced restoration	Active revegetation	Reintroduction of peatland vegetation, including seeding or transfer of characteristic species, e.g. *Sphagnum* mosses, to restoration sites. Moss Layer Transfer Technique is an example which includes preparatory steps.	Pouliot et al. (2012); Putkinen et al. (2018); González & Rochefort (2014, 2019); González et al. (2014)

Table 2. Model description and outputs – normal error structure applied across all tests

Test	**Model description**			**Model outputs**						
	Response variable	Fixed effects and interactions	Random effects	Chi square	P value	Sample size	Categories	Post-hoc testing		
								95% confidence level		
								Mean	**LCL**	**UCL**
(a) Testing the effect of peatland status (degraded, restoration, and intact sites)	*Sphagnum* spp. cover[1]	Peatland status	Study ID	691.83	<0.0001 ***	688	*Degraded*	24.99	23.78	26.20
						3,611	*Restoration*	35.23	34.80	35.67
						1,532	*Intact*	117.66	116.40	118.92
	Sphagnum species richness			210.50	<0.0001 ***	592	*Degraded*	1.01	0.92	1.12
						3,533	*Restoration*	2.12	2.08	2.16
						1,605	*Intact*	2.60	2.56	2.64
	Peatland specialists cover[1]			399.52	<0.0001 ***	579	*Degraded*	43.28	42.40	44.18
						3,389	*Restoration*	50.35	49.86	50.84
						829	*Intact*	104.82	103.75	105.88
	Peatland specialists species richness			267.55	<0.0001 ***	592	*Degraded*	2.62	2.41	2.83
						3,533	*Restoration*	4.75	4.68	4.82
						1,605	*Intact*	5.41	5.33	5.50
								Slope (model estimate ± SE)		
(b) Testing the effect of monitoring age[2]	*Sphagnum* spp. cover[1]	Restoration age	Study ID	516.1	<0.0001 ***	5,831		0.0264 ± 0.00112		
	Sphagnum species richness			207.18	<0.0001 ***	5,730		0.0838 ± 0.00574		
	Peatland specialists cover[1]			471.5	<0.0001 ***	4,797		0.0305 ± 0.00136		
	Peatland specialists species richness			268.1	<0.0001 ***	5,730		0.152 ± 0.00910		
(c) Testing the effect of interventions (no intervention, stressor removal, and active revegetation) on restoration trajectories[2]	*Sphagnum* spp. cover[1]	Intervention level * monitoring age	Study ID	45.478	<0.0001 ***	2,193	*Passive*			
						1,055	*Basic*			
						2,583	*Enhanced*			
	Sphagnum species richness			66.163	<0.0001 ***	2,179	*Passive*			
						969	*Basic*			
						2,582	*Enhanced*			
	Peatland specialists cover[1]			39.227	<0.0001 ***	1,388	*Passive*			
						939	*Basic*			
						2,470	*Enhanced*			
	Peatland specialists species richness			102.65	<0.0001 ***	2,179	*Passive*			
						969	*Basic*			
						2,582	*Enhanced*			

[1] Arcsine square root transformed [2] Intact sites excluded as they are not associated with suitable temporal data for the purposes of this modelling

Figures

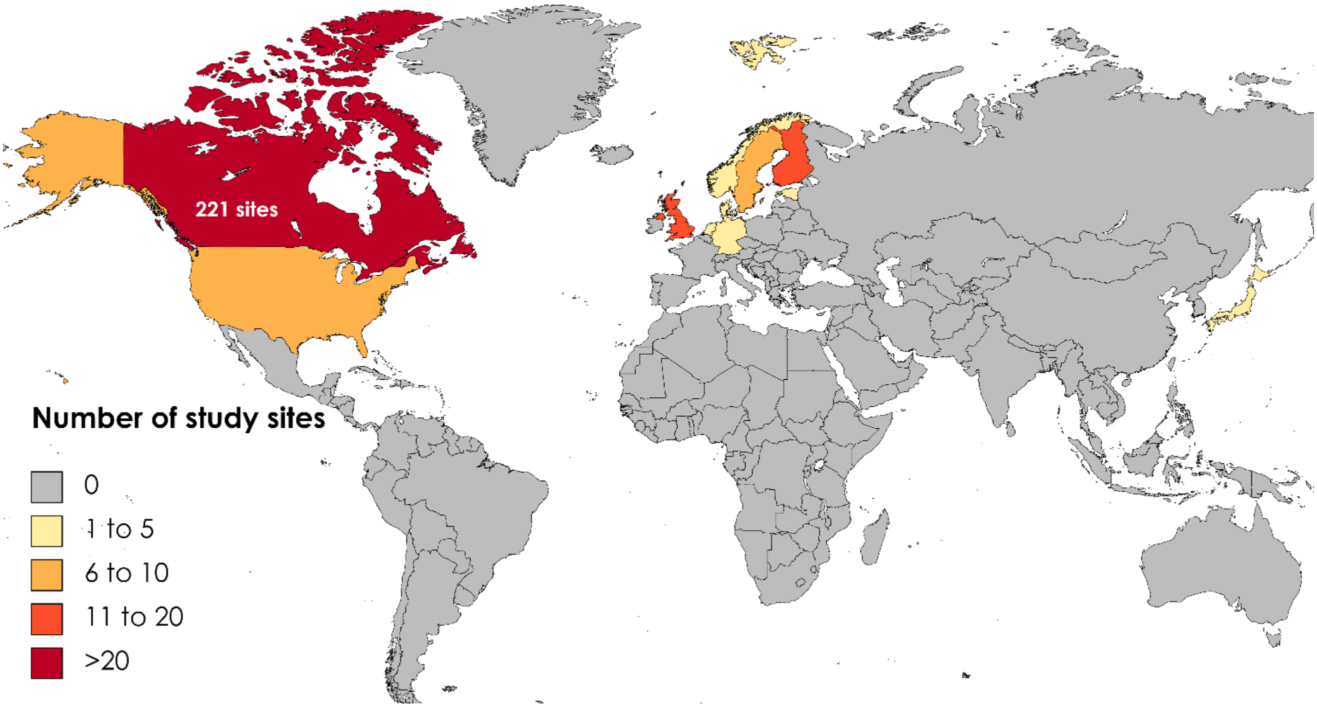

Figure 1. Number of study sites per country

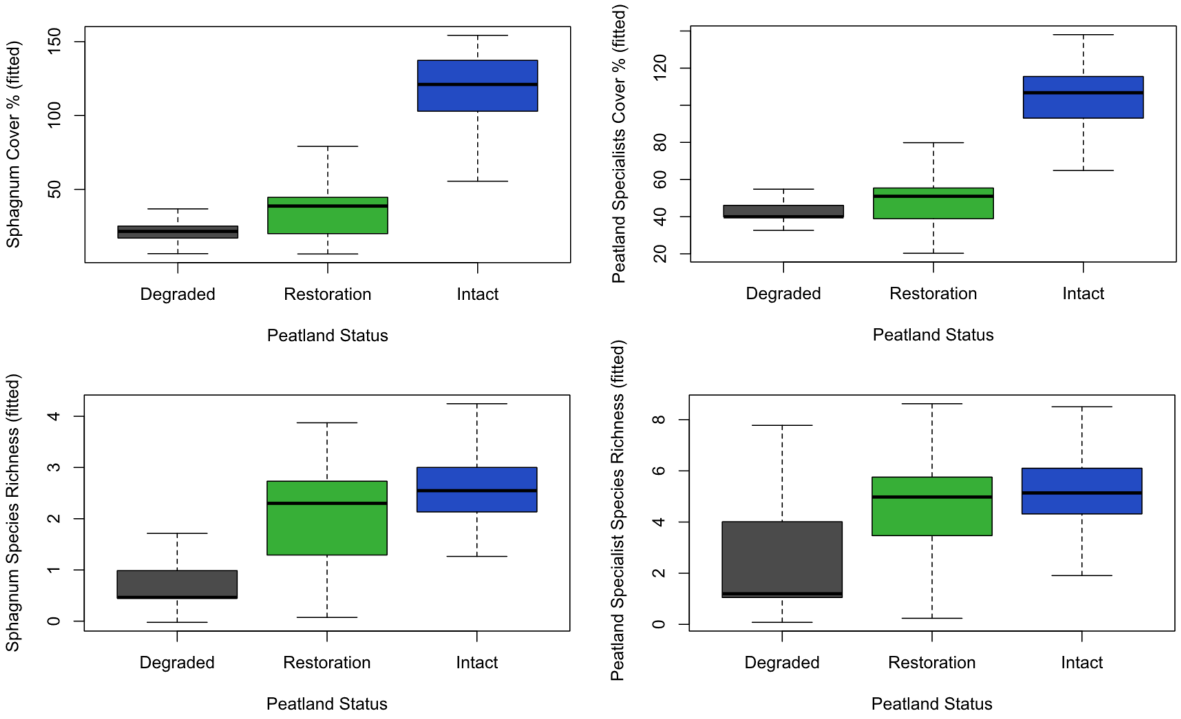

Figure 2. Model-predicted cover of *Sphagnum* (a) and peatland specialists (b), and species richness of *Sphagnum* (c) and peatland specialists (d) for intact, restoration, and degraded peatlands.

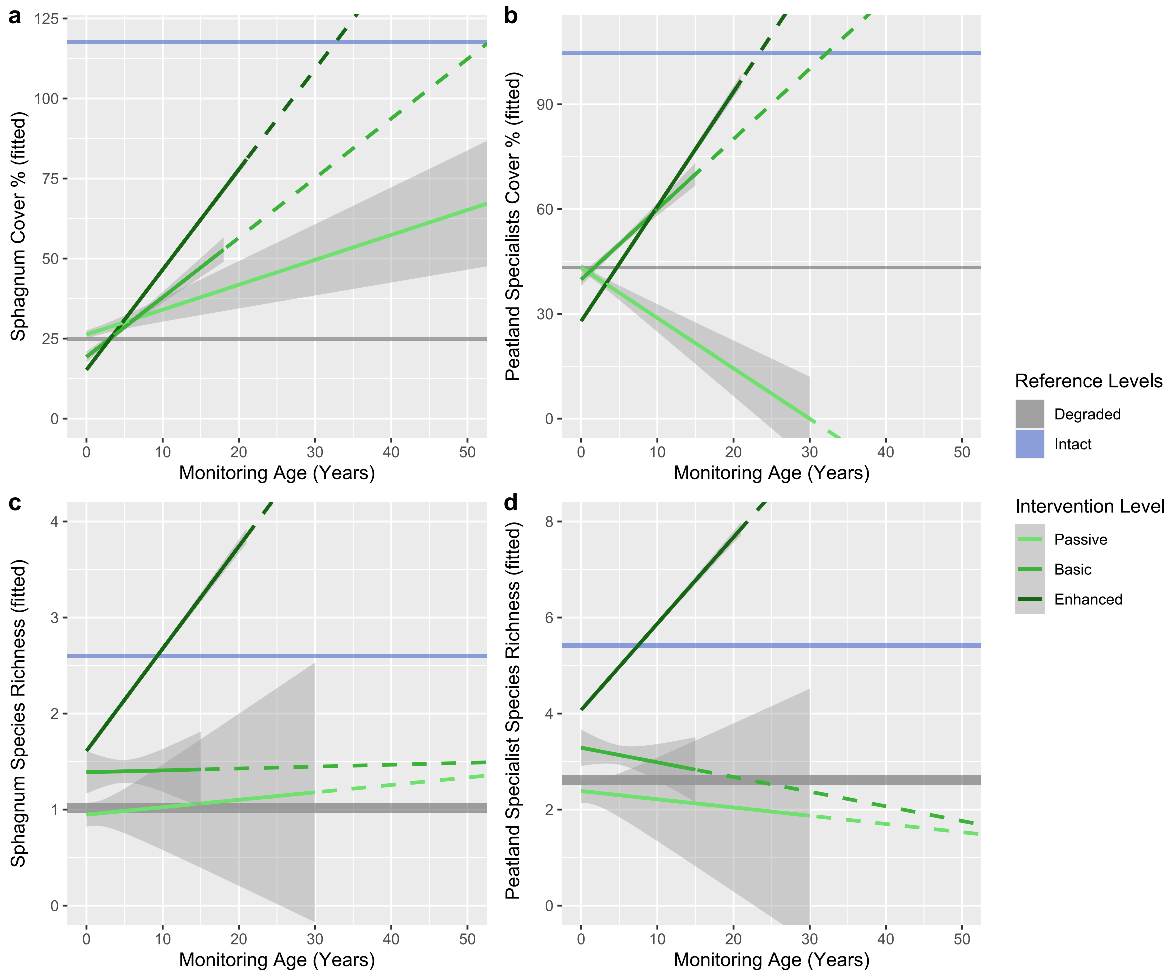

Figure 3. Model-predicted response trajectories of cover for *Sphagnum* mosses (a) and peatland specialists (b); species richness for *Sphagnum* mosses (c) and peatland specialists (d). Dashed lines indicate projected trajectories outside range of study years assuming comparable rate of change; ribbons indicate 95% confidence intervals; reference levels are the means ± standard error for degraded and intact peatlands.