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Assessing the effects of peatland restoration

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I Summary

Naturally functioning peatlands, referred to as 'mires', are of high biodiversity value and fulfil a wide range of ecosystem functions, such as landscape-scale water regulation or the long-term storage of carbon and nutrients. Due to drainage, mainly for agriculture and forestry, more than 95% of German peatlands are degraded, resulting in the loss of biodiversity and the vital ecosystem functions of mires. Therefore, in a time of anthropogenic climate change and biodiversity loss, restoration efforts have increased in the past few decades, and international programmes and policies have progressively recognised the importance of peatland restoration. A central part of restoration is monitoring to evaluate success, inform on restoration outcomes and adapt restoration practices. However, the monitoring of peatland restoration is not standardised, and a methodology to assess mire-specific biodiversity is missing. Raising the water table, re-establishing peat accumulation and peatlands as carbon sinks, enhancing mire-specific biodiversity and saving GHG emissions are central goals of peatland restoration, but collectively analysed monitoring data is scarce. Although peatland restoration was found to enhance peatland ecosystem functioning and biodiversity, the outcome is not comparable to pristine mires, and factors influencing restoration success remain unclear.

Therefore, this thesis presents the results of three studies:

In **Study I**, a user-friendly, robust indicator system to assess mire-specific biodiversity is developed for north-east Germany, encompassing the different levels of mire-specific biodiversity and enabling an overall assessment. Value scales to rate mire-specific biodiversity in six categories are derived based on data from 30 study sites. To ensure robustness, the indicator system assessment is compared with assessments made by experts and practitioners. The comparison shows high correspondence. Furthermore, a practical example is given, demonstrating the indicator system's use for restoration projects.

In **Study II**, the results for total organic carbon content (TOC), total nitrogen content (TN), the C/N ratio, pH value and dry bulk density (BD) sampling, as well as a description of the structure of near-surface peats in six restored fens in north-east Germany before (2002–2004) and after (2019–2021) restoration, are presented. Following successful rewetting with a water table at the surface, TOC increased comparable to values of undisturbed peats. In addition, the C/N ratio increased, meaning a decrease in nutrient availability. BD decreased, and values of sedge-dominated sites were similar to undisturbed peats. The PH value remained stable over time. A new, slightly decomposed peat moss layer on top of the degraded peat in peat moss-dominated sites was determined, as well as a structural change in the degraded near-surface peat in sedge-dominated peatlands to a more homogenous sludge mass with larger re-aggregates.

In **Study III**, the effects of peatland restoration on the water table, peat accumulation, mire-specific biodiversity and GHG emissions of 33 forest peatlands in north-east Germany are described. In peatlands drained by ditches, hydraulic restoration led to a rise in the water table and the peat-accumulating area, increased mire-specific biodiversity, and decreased estimated GHG emissions. Still, the highest positive values in all analysed parameters were rarely determined. In contrast, peatlands, not drained by ditches but showing early signs of degradation, were restored by management measures – in this context the removal of tall vegetation or upcoming trees, as well as forest restructuring in the peatland catchment area. These peatlands were in a better initial state and management measures did not significantly enhance but instead preserved pre-measure conditions. In general, groundwater-abstracting facilities in the peatland catchment area hindered restoration success. Principal component analysis showed that the acid-base-ratio and tropic conditions of the peatland before restoration, hydrogenetic mire type and peatland size were not correlated with restoration success. In contrast, years since restoration had a positive relation.

By using an easily applicable monitoring methodology, including the newly developed indicator system, to assess mire-specific biodiversity, the effects of peatland restoration on central restoration aims were robustly evaluated.

The results of this thesis highlight the importance of preserving all remaining mires, as restoring peatlands to near-natural conditions is rarely achieved in the short term. However, as peatland rewetting especially quickly raises the water table, leading to peat accumulating conditions, TOC values in near-surface peats comparable to values of undisturbed peats, estimated savings in GHG emissions and greater habitat heterogeneity, it is necessary to rewet all degraded peatland as one key factor in mitigating climate change and fighting biodiversity loss. Mire-specific species increase slightly and not significantly after restoration, but the determined lower nutrient availability is assumed to favour re-colonisation by mire-specific species in the long term. Management measures preserve pre-measure conditions within the study's time frame. As the positive effects of forest restructuring in peatland catchment areas are estimated to be long-term, this approach should be implemented immediately and considered for still functioning mires in the context of reduced water availability due to climate change. The decrease in BD and the newly accumulated peat layers in peat moss-dominated peatlands indicate the development of a new acrotelm, an essential factor in restoring peatlands to self-regulating mires.

In conclusion, peatland restoration enhances ecosystem functionality and biodiversity, thus enhancing peatlands' resilience against the negative influences of global warming, mitigating climate change, and fighting biodiversity loss.

1. Introduction

1.1. Research background and gap

Peatlands have been increasingly recognised by society and politics in the past decades as one key factor in climate change mitigation and the conservation of biodiversity (Convention on Wetlands 2021; Tanneberger et al. 2021). Nations and actors worldwide are urged by the United Nations Environment Assembly, through the resolution on "Conservation and Sustainable Management of Peatlands" (UNEA 2019), to place greater emphasis on conserving, sustainably managing and restoring peatlands, in recognition of them as a major global carbon (C) store and for their provision of vital ecosystem functions and biodiversity value. Peatlands are one of eight ecosystem categories focused on within the United Nations Decade on Ecosystem Restoration (2021-2030), and their protection and restoration are important elements in meeting the targets of the Global Biodiversity Framework of the Convention on Biological Diversity (CBD) (UN 2020; CBD 2022). Preventing, halting and reversing peatland degradation is classed as a nature-based solution to meet the goals of the United Nations Framework Convention on Climate Change (UNFCCC) and the Paris Agreement (Fuchs & Noebel 2022).

1.1.1. Mires – ecosystem functions and biodiversity

These and other policy and action programmes arise from the several ecosystem properties and functions of peatlands, especially mires (cf. Kimmel & Mander 2010). The term 'peatland' refers to an area with a naturally accumulated layer of peat at the surface, while the term 'mire' addresses naturally functioning peatlands in which peat is currently being formed and accumulated (Joosten et al. 2017).

In mires, based on water-saturated conditions in the soil due to a water table close to the surface, plant remains of the mostly highly adapted, peat-forming vegetation is only partially decomposed, and organic matter accumulates as peat (Parish et al. 2008). This sedentarily accumulated material is defined by at least 30% dry mass of dead organic matter (Joosten et al. 2017). Through this process, high amounts of C and nutrients are removed from the matter cycle, as water-saturated conditions and the filtering characteristics of the peat result in the conversion of C, nutrients as well as pollutants or their sequestration in the newly accumulated peat (Joosten & Clark 2002; Luthardt & Wichmann 2016; Price et al. 2016). Their function as a C-sink has led to an estimated global C store of over 600 gigatons, which accounts for 44% of all soil C (IUCN 2021), making peatlands a significant C store and an important regulator of the global climate (Joosten & Clarke 2002; Kimmel & Mander 2010).

Furthermore, mires fulfil diverse hydrological functions. Depending on their location and integration within the landscape, they can act as flood retention areas, water storage basins or groundwater nourishment areas (Luthardt & Wichmann 2016). The mires' catchment hydrology is regulated by water storage, groundwater recharge and discharge (Joosten & Clarke 2002; Kimmel & Mander 2010). By filtering inflowing water, mires improve water quality and regulate catchment hydrochemistry, benefitting, for example, the quality of drinking water supplies or adjacent water bodies (Price et al. 2016). In addition, compared to mineral soils, they have 10 - 15% greater evaporation capacity, and in hot periods, the release of dew and water vapour can have a cooling effect, which has a stress-reducing outcome on the surrounding landscape (Luthardt 2014; Luthardt & Wichmann 2016).

In addition to the abovementioned regulating functions, mires provide production and informational functions, for example, in producing wild plants, animals and woodland or as places for recreation, tourism and spirituality (Kimmel & Mander 2010).

This variety of ecosystem functions is closely linked to the biodiversity of mires (Bonn et al. 2016; Luthardt & Wichmann 2016), which is highly heterogenic on all levels (Minayeva et al. 2017). Although all mire ecosystems are characterised by permanent water saturation and the formation of peat (Parish et al. 2008), they range highly in terms of genesis, geomorphology, hydrology and ecology (Joosten et al. 2017a). Furthermore, they vary in physiognomy (morphological forms) and structural heterogeneity on different scales. Moen et al. (2017) describe 13 mire massif types in 10 mire regions and 52 sub-regions for Europe alone. On a smaller spatial unit, Minayeva & Sirin (2012) describe the structural diversity for microtopes (e.g., hummock-hollow-complex), macroforms (e.g., hummock) and microcenosis (e.g., an element of the vegetation mosaic). The often rare and endangered species found in mire areas are highly specialised, as they need to be adapted to water-saturated conditions and mostly extreme nutrient and pH values (Minayeva et al. 2008; Littlewood et al. 2010; Prentice 2011; Aapala et al. 2014). Although rarely researched, a wide range of genetic diversity has been verified, e.g., for *Drosera* and *Sphagnum* species (Shaw et al. 2008; Yousefi et al. 2019; Eschenbrenner et al. 2019).

1.1.2. Peatlands – distribution, degradation and consequences of drainage

Mire biodiversity is present in peatland ecosystems worldwide, which are widely but not homogenously distributed (Moen et al. 2017; Xu et al. 2019). The total peatland area is estimated to be 4.23 million km^2 globally (Xu et al. 2018), and although roughly 80% of this is still in a natural state, 20% has been drained for agriculture, forestry, peat extraction and infrastructure. Natural peatlands are located in secluded parts of Canada, Alaska and Siberia, whilst the drained sites are mainly situated in the temperate zone and the (sub)-tropics (Joosten 2016). Currently, 594,018 km^2 of peatland area is present in Europe (Joosten & Tanneberger 2017), occurring in all countries but with a higher existence in the north than in the south (Moen et al. 2017). 50% of the European Union peatland area is degraded (Tanneberger et al. 2021), with over 95% of the total 1,280,000 ha in Germany, for instance, in this condition (GPD 2017). In total, 72% of German peatlands are drained for agriculture, 14% for forestry, 7% for infrastructure, 1.5% for peat extraction and 1.5% for other land use types. This leaves only 4% of unused areas or those under nature conservation (Trepel et al. 2017). The main distribution of peatlands within Germany is found around the northern states of Schleswig-Holstein, Lower Saxony, Mecklenburg-Vorpommern and Brandenburg (Succow und Joosten 2001). An evaluation in some states (Lower Saxony, Schleswig-Holstein and Brandenburg) shows that 30% of peatland area has been lost compared to 50-60 years ago, mainly due to agricultural use and subsidence (Bauriegel 2014; Burbaum & Filipinski 2015; Trepel et al. 2017). In the study area (Brandenburg, north-east Germany), the usage of peatlands, mainly for agriculture and forestry, has reduced the total area in Brandenburg from 270,000 ha (based on the Prussian geological map) to 163,000 ha in 2013 (LfU 2016).

Drainage results in the loss of ecosystem functions and the biodiversity of mires.

Is the water table drained below surface, peat formation is interrupted and secondary pedogenetic processes, namely consolidation, compaction, shrinkage, humification and mineralisation alongside dislocation, leaching and the accumulation of soil substances, are initiated. The resulting peat subsidence of up to 25 mm yearly often leads to deeper drainage that lowers the water table even further, therefore leading to ongoing, ever-deepening peat degradation and loss (Zeitz 2016).

These processes in particular reduce the regulating functions of mires (Luthardt & Wichmann 2016); for instance, climate regulation changes from a 'cooling' effect via the long-term storage of C to a 'heating' effect caused by ceasing C sequestration and the release of formerly stored C as carbon dioxide (CO_2) and nitrous oxide (N_2O) (Drössler et al. 2014; Joosten et al. 2016). In Germany, peat soils caused emissions of 53.7 Mio t CO_2-eq. in 2021, accounting for 7% of total annual GHG emissions. Most of these emissions, 79.8% or 42.8 Mio t CO_2-eq., result from the agricultural use of drained peat soils (UBA 2022). In the study area, i.e., Brandenburg, the total amount of GHG emissions from peat soils is estimated at 6.3 Mio t CO_2-eq. per year, accounting for 11% of the state's annual GHG emissions (Reichelt 2021).

The hydrological functioning of peatlands is negatively affected by drainage (Gebhardt et al. 2012; Zeitz 2016). As a result of soil structure changes, the water retention capacity of the degraded peat is lowered, thereby leading to a hydrophobic topsoil and, therefore, for example, impeded landscape water regulation, increased flood risk or limited storage capacity of plant available water (Zeitz 2016; Word et al. 2022). Peatland drainage leads to a complete change in nutrient cycling, which in turn results in severe consequences for water quality regulation. Whereas a mire actively stores C, nitrogen, sulphur and toxic substances, and shows only moderate exports of dissolved organic carbon (DOC), organic acidity, particulate organic carbon (POC) and nutrients, severely drained and disturbed areas export high amounts of DOC, POC, sulphate (SO_4), nitrate (NO_3), toxic substances and acidity (Price et al. 2016). This export negatively affects inter alia adjacent water bodies, leading to hydrochemical and ecological impacts, such as changing from an oligo to an eutrophic lake or increasing algal biomass (Kondelin 2006).

With respect to biodiversity, the literature often refers to the loss of species following drainage – a phenomenon reported worldwide, e.g., the reduction in *Sphagnum* cover and other peat-forming species in the northern Andes (Benavides 2014), the decline of the Northern European bird population (Fraixedas et al. 2017) or the negative effects on species richness and the abundance of *Odonata* species of bogs in Finland (Elo et al. 2015). On an ecosystem level, the International Union for Conservation of Nature lists 11 out of 13 mire biotopes as "threatened," 23% of which is "endangered" or "critically endangered", and 62% is vulnerable (Jannsen et al. 2016). In Brandenburg, 62% of mire-specific vascular plants have a Red-List Status of "extinct," "at risk of extinction" or "highly endangered" (cf. LUA 2006; Luthardt 2014c; Hammerich et al. 2022).

Next to drainage, anthropogenic climate change is already influencing – and will continue to affect – peatland hydrology and soil temperature with consequences for their ecology, biochemistry, biodiversity and interactions with the earth system (Gallego-Sala et al. 2016; Antala et al. 2022; Loisel & Gallego-Sala 2022). Climate change will cause a further increase in temperature with extreme weather events, such as intensive precipitation events and droughts and global warming is more likely than not to reach 1.5 °C in the near term (IPCC 2023). In Brandenburg, the average temperature has increased by 1.3 K between 1881 and 2018 (DWD 2019). Negative impacts affecting peatlands in Brandenburg are expected to include decreasing seepage water rates, and thus lower groundwater recharge, the drying out of the upper soil layers and increased peat mineralisation as well as changes in vegetation (Schulz-Sternberg et al. 2009; Nusko & Luthardt 2014).

1.1.3. Peatland restoration

To stop the loss of mire biodiversity and ecosystem functions, conservation started over 100 years ago, moving slowly from local initiatives, to national strategies and then to international programmes and conventions (Kaakinen & Salminen 2006; Tanneberger et al. 2017). Peatland restoration is a far newer activity, beginning in the late 20th century by initially restoring biodiversity components and later ecosystem functions, with increasing experience worldwide (Bonn et al. 2016a; Tanneberger et al. 2017). Generally, the ecological restoration of ecosystems is the "process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (SER 2004). In respect to peatlands, a differentiation is made between restoration and rewetting. While restoration aims at turning the peatland back as closely to its natural condition before human-induced degradation, rewetting aims at raising the water table to a peat-accumulating state; the latter option is always the first step towards restoration (GMC & Wetlands International 2023).

Peatland restoration is often carried out by reconstructing the hydrological regime and raising the water table through hydraulic measures such as blocking drainage systems. Furthermore, techniques such as topsoil extraction, removing non-native trees or reintroducing typical vegetation are commonly used (Gonzáles et al. 2014; Pfeifenberger & Fock 2015; Bonn et al. 2016).

The need to conserve and restore peatlands is now internationally recognised, among others, in the Ramsar-Wetland-Convention, the CBD, the UNFCCC and the Common Agricultural Policy of the European Union. On a national German basis, since 2022/2023, the need has been recognised in the German Peatland Protection Strategy (*Nationale Moorschutzstrategie*) and the Action Programme for Natural Climate Protection (*Aktionsprogramm Natürlicher Klimaschutz*). The Convention on Wetlands (2021) estimates that rewetting all drained peatlands, with a global area covering 50 million ha, is necessary to meet the objectives of the UNFCCC and the Paris Agreement. How much has already been rewetted or restored, however, is unclear. For example, Indonesia rewetted 300,000 ha in 2021, targeting 1.2 million ha by 2024 as part of its restoration programme for tropical peatlands (Jong 2022). In Germany, it is estimated that since 1980, 70,000 ha have been rewetted (Abel et al. 2019). The National Environmental Ministry is financially supporting four projects, running over the next ten years, to restore and sustainably use peat soils (Bundesregierung 2022). Additionally, the Federal Ministry of Food and Agriculture is funding demonstration projects on paludiculture, namely the use of wet peatland for agriculture or forestry, with approximately €10 million per year for ten years (BMEL 2023). In Brandenburg, the 'WetNetBB – Management and biomass utilisation of wet fens: Network of model and demonstration projects in Brandenburg's peatland regions' project is funded within this programme. A further local example is the 'Protection programme for forest peatlands in Brandenburg (W*aldmoorschutzprogramm Brandenburg*)', currently running at around 120 restoration sites (Hammerich et al.

2022b). Nevertheless, political and restoration activities are presently insufficient in this regard and are progressing very slowly (Roe et al. 2019; GMC & Wetlands International 2022).

1.1.4. Monitoring peatland restoration – current state of research, challenges and gaps

In order to ensure success in future peatland restoration activities, it is essential to monitor projects and transfer lessons learned to other restoration schemes (Rochefort & Andersen 2017; Convention on Wetlands 2021). Although peatland restoration has developed from trial and error to more knowledge-, experience- and scientific-based practices, many research gaps still exist (Chimner et al. 2016; Tanneberger et al. 2017). A wide range of monitoring studies focus on the development of different peatland properties and functions following restoration, varying from single to multiple case studies (e.g., Mälson et al. 2008; Laine et al. 2011; Mrotzek et al. 2020; Kreyling et al. 2021).

In general, peatland properties react in different timelines following restoration. Rises in the water table, and thereby savings in CO_2 emissions, are usually achieved in the short term, but changes in vegetation in the medium to long term require longer monitoring timelines (Haapalehto et al. 2011; Aapala et al. 2014a; Kareksela et al. 2015; Renou-Wilson et al. 2018). Drainage and degradation intensity have resulted in diverse conditions of peatlands from near-natural and undisturbed to non-natural, highly degraded, and restored sites (cf. Wagner & Wagner 2004). Therefore, peatland conditions, characteristics, applied restoration measures, and the duration after rewetting, which supposedly influence restoration success, vary highly between sites. Monitoring data, which is often collected for individual sites, has rarely been analysed collectively (cf. Bonnett et al. 2009; Grand-Clement et al. 2013; Renou-Wilson et al. 2018). Existing data in this regard is often not published, and the availability of long-term monitoring information, compared to restoration projects, is rare and non-standardised. Additionally, robust monitoring schemes for peatland ecosystem functions and biodiversity are lacking (Andersen et al. 2016).

Regardless of the type of peatland, restoring their hydrologic regime is the basis for a successful restoration project (Chimner et al. 2016). Still, unstable and low water tables, due to anthropogenic changes, seem to be the central challenge in this endeavour (Convention on Wetlands 2021). Many studies report a successful rise in the water table after restoration (Haapalehto et al. 2011; Laine et al. 2011), but the effects are not always comparable to pristine sites, with larger water table amplitudes in rewetted sites and only successful rewetting of certain – often central – lower areas within the peatland (e.g., Strobel et al. 2019; Krejčováo et al. 2021; Kreyling et al. 2021).

The new formation of the 'acrotelm', namely highly dynamic near-surface peat, which can actively store water (Tolonen & Turunen 1996), is seen as a critical challenge in successfully restoring peatlands to self-regulating mires (Convention on Wetlands 2021). Whereas changes in peat properties following drainage are widely researched and understood (e.g., Stegmann & Zeitz 2001; Gebhardt et al. 2009; Zeitz 2016), the processes involved in recovering formerly degraded peat and the accumulation of new peat after rewetting, are only beginning to be comprehended (Chimner et al. 2016; Word et al. 2022). This is especially the case for non-moss peats (Kotowski et al. 2016; Michaelis et al. 2020; Mrotzek et al. 2020) compared to *Sphagnum* peats (e.g., Rochefort et al. 2003; Rochefort 2009; Kareksela et al. 2015; Gaffney et al. 2020; Rochefort & Andersen 2017). As reported by Strobel et al. (2019), when a favourable water table is achieved after rewetting, better water-holding capacity and lower decomposition in near-surface peats occurred in *Sphagnum*-dominated peatlands. Kareksela et al. (2015) found comparable growth rates of *Sphagnum* peat surface layers to pristine sites within 5 and 10 years of rewetting, albeit C sequestration was lower than in pristine sites. Word et al. (2022) observed higher microporosity and lower macroporosity alongside key water retention parameters in formerly degraded, non-moss peats after rewetting and high water saturation frequency, indicating potential recovery to pre-disturbed conditions. A 'proto-peat', consisting of newly deposited material (roots, radicals, litter), is described by Michaelis et al. (2020) and Mrotzek et al. (2020) for fens in north-east Germany, indicating the recovery of peat-accumulating systems within 20 years of restoration. Additionally, values for standard soil properties for describing the state of peat soil, namely total organic carbon (TOC), total nitrogen content (TN), C/N ratio, bulk density (BD) and structural description, exist for natural and degraded peats in north-east Germany and comparable regions (e.g., Succow 1988; Grosse-Braukmann 1990; Naucke 1990; Loisel et al. 2014) but barely for restored peats (Mrotzek et al. 2020). This leaves a gap in estimating the recovery of a functioning near-surface peat layer after restoration, inter alia, based on key soil properties.

Mires are characterised by a high level of maturity, due to their longevity and, therefore, resilience against natural disturbances (Meier-Uhlherr et al. 2014; Loisel & Gallego-Sala 2022; Luthardt et al. 2023). Once this continuity is disturbed to a certain degree, however, their natural functioning and biodiversity become challenging in terms of restoration (cf. Loisel & Gallego-Sala 2022). Concerning mire biodiversity, most research focuses on the development of vegetation or specific fauna following restoration. It has been shown that although plant species favouring wet conditions seem to recover after water table rises following restoration, mire-specific species appear to be absent within the studied timeframes and only slowly 'move back' (Haapalehto et al. 2011; Laine et al. 2011). The plant community composition of rewetted fens is dissimilar to pristine sites (e.g., Krejčováo et al. 2021), whilst a trend towards the dominance of tall, graminoid wetland plants (Kreyling et al. 2021) and a more homogenous species composition, dominated by a few high-coverage species (Mälson et al. 2008), has been observed. Overall, assessments of mire biodiversity are rare, and only a few examples are found in multi-indicator assessment approaches (Görn & Fischer 2011; Joosten et al. 2015; Strobel et al. 2019). As such, a standardised, cost-effective, practitioner-friendly and reproducible methodology to assess mire-specific biodiversity is missing.

Currently, the most frequently researched field in peatland science is quantifying GHG emissions from degraded, near-natural and restored peatlands (Van Bellen & Larivière 2020). Whereas degraded peatlands emit CO_2 and N_2O (Drössler et al. 2014; Joosten et al. 2016), mires are considered moderate to climate-neutral net GHG sinks (Barthelmes et al. 2015; Humpenöder et al. 2020). Shallow water bodies following restoration, especially in the first few years, lead to high methane emissions, which counteract savings in CO_2 and N_2O (Zak et al. 2015), albeit peatland restoration is estimated to generally lead to savings in GHG emissions compared to the previously drained state (Wilson et al. 2016; Humpenöder et al. 2020). However, emission reductions vary over time and in different climatic regions and are therefore difficult to generalise. Furthermore, measuring GHG flux data is costly and requires expert knowledge. Using an easily applicable approach, such as the Greenhouse Gas Emission Site Type (GEST) model by Couwenberg et al. (2011), to estimate GHG data of restoration projects has rarely been done (cf. Herrmann et al. 2018; Jarašius et al. 2022; Martens et al. 2022), and collectively analysed data in the context of other restoration goals is missing.

1.2. Research Aim

This thesis aims to contribute an easily applicable methodology for monitoring peatland restoration, apply this methodology on peatland restoration cases and then use the results to learn for future restoration projects. It therefore attempts to close the knowledge gap in terms of measuring mire-specific biodiversity and contribute experience to the development of water tables, peat properties and peat accumulation, mire-specific biodiversity and estimated GHG emissions following restoration, as well as determine possible factors influencing restoration success. The studies herein were carried out in Brandenburg, north-east Germany.

Within this framework, specific research questions were set and investigated across three different studies:

Study I: Hammerich J, Dammann C, Schulz C, Tanneberger F, Zeitz J, Luthardt V. (2022) Assessing mire-specific biodiversity with an indicator based approach. *Mires and Peat* 28/32:1–29 DOI: 10.19189/MaP.2021.SJ.StA.2205

a. Which indicators, attributes and measure values are suitable for assessing mire-specific biodiversity on different levels?
b. Which value scales for each indicator represent the degree (none to very high) of mire-specific biodiversity?
c. Is the assessment via indicators comparable to the assessment by experts and practitioners in peatland restoration?

Study II: Hammerich J, Schulz C, Probst R, Lüdicke T, Luthardt V (2024) Carbon content and other soil properties of near-surface peats before and after peatland restoration. *PeerJ (in press)*

a. How do TOC, TN, C/N ratio, pH value, BD and structure – as important indicators in the recovery of near-surface peat and carbon sink function – change, from before to after restoration?
b. Are the results comparable to pristine site values (based on a comparison of the literature)?

Study III: Hammerich J, Schulz C, von Wehrden H, Zeitz J, Luthardt V (n.d.) Monitoring peatland restoration in forests – The effects of hydraulic and management measures on the water table, peat accumulation, mire-specific biodiversity and greenhouse gas emissions. *Restoration Ecology (submitted April 2024)*

a. Which easy applicable monitoring methodology can be used to assess changes in the water table, peat-accumulating area, mire-specific biodiversity and GHG emissions, before and after peatland restoration?
b. How, and to what extent, do water tables, peat-accumulating areas, mire-specific biodiversity and GHG emissions change from before to after restoration?
c. How do restoration measures, groundwater-abstracting facilities in the peatland catchment areas, trophic conditions, the acid-base ratio before restoration, hydrogenetic mire type, peatland size and years since restoration influence restoration success?

An overview on the studies and their linkages is shown in Figure 1.1.

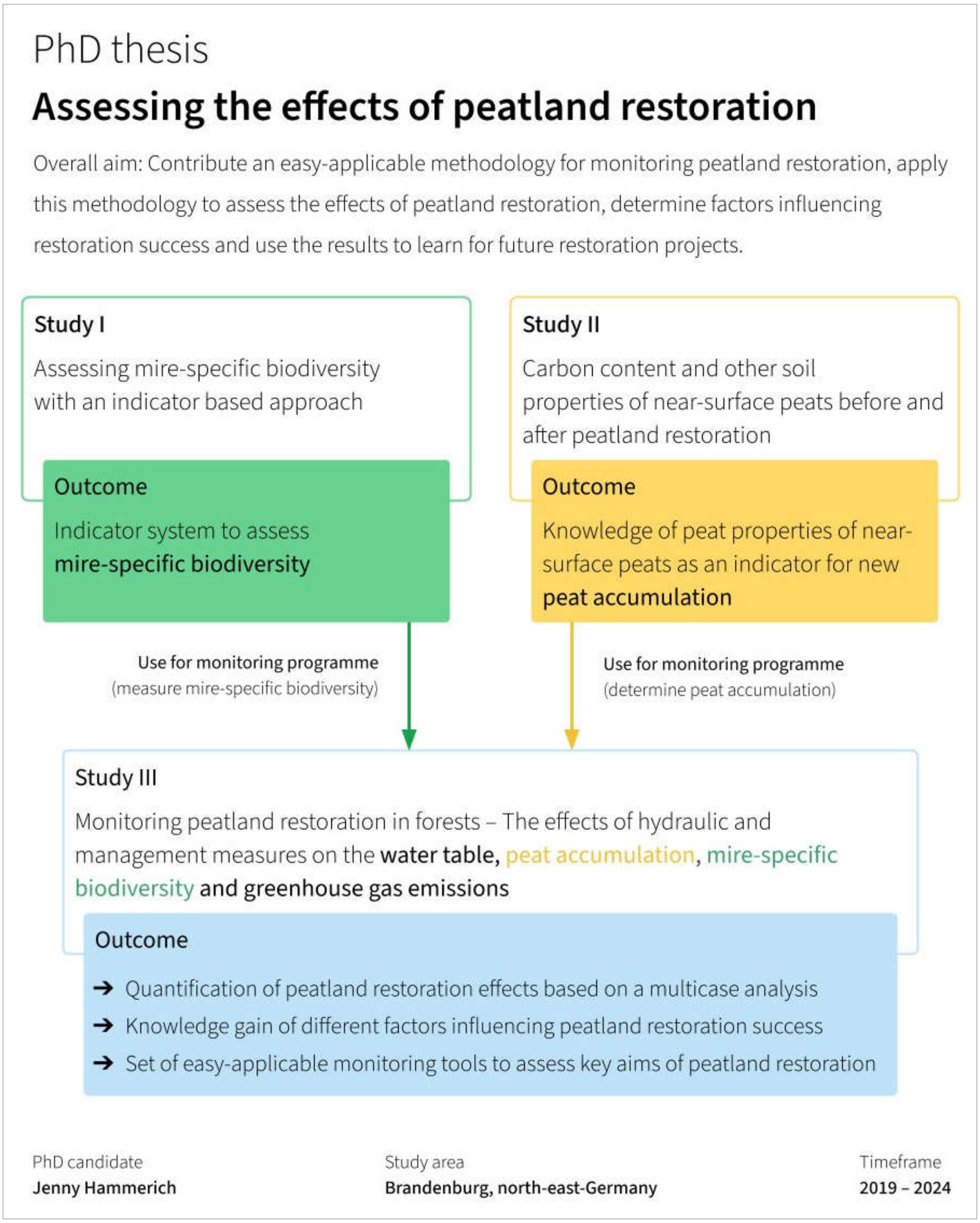

Figure 1.1: Different studies and their linkages of the PhD-Thesis "Assessing the effects of peatland restoration" by Jenny Hammerich

2. Study I: Assessing mire-specific biodiversity with an indicator based approach

2.1. Authors and affiliations

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2.2. Abstract

The biodiversity of mires is characterised by a small number of highly specialised species, mostly high spatial heterogeneity and a strong influence of abiotic factors such as high water table and soil substrate (peat). To assess mire-specific biodiversity, indicators that represent and value all of these characteristics are needed. In this study, we present a system of such indicators for the example of north-east Germany. Our indicators encompass different levels of mire-specific biodiversity and enable an overall assessment. We place special emphasis on high user-friendliness. The attributes considered have been well researched in the study area. Based on data from 30 study sites, we developed scales for rating mire-specific biodiversity in six categories. To evaluate the indicator system, we compared the assessment of selected peatlands via the indicator system with the assessments of experts and practitioners in peatland research and management. This evaluation showed high correspondence. We also demonstrate the use of the indicator system as a practical tool for assessing the effects of peatland restoration, and provide suggestions for its application in other geographical regions.

2.3. Key words

Biocoenosis, ecosystem, monitoring, peatland, species, value, vegetation

2.4. Introduction

Biodiversity is the complex of diversity within species, between species, and of biocoenosis within their habitats, i.e. ecosystems (UN 1992; Küchler-Krischung & Walter 2007; Wittig & Niekisch 2014). It has gained global importance owing to its drastic decline. In recent decades, the effort to maintain existing biodiversity and restore lost biodiversity has been driven by various international treaties (e.g. Convention on Biological Diversity, Ramsar Convention, Bonn Convention, etc.) and their national implementations as well as by an increasing number of biodiversity conservation projects. This creates an immediate need for assessment methods to quantify biodiversity loss or gain in general, and to judge the success of privately or publicly funded restoration projects in particular. However, because biodiversity is complex, methods which are not only validated but also cost effective and practitioner friendly are scarce (Brunbjerg et al. 2018).

Peatlands (areas with a naturally accumulated layer of peat) and in particular mires (peatlands with a vegetation that forms peat) (Joosten et al. 2017) are of great and often unrecognised biodiversity value (Parish et al. 2008; Prentice 2011; Minayeva et al. 2017). The biodiversity of mires is the key element for a wide range of ecosystem services, such as landscape-scale water regulation as well as nutrient and carbon storage (Bonn et al. 2016; Luthardt & Wichmann 2016). Nonetheless, around 10 % of the former total peatland area in Europe has been lost due to a long history of drainage for agriculture and other land uses. Forty-eight percent of the remaining European peatland area is degraded, in Germany only 4 % of all peatlands are unused and/or nature conservation areas (Joosten & Tanneberger 2017; Trepel et al. 2017). Of the 163,150 ha of peatlands remaining in the federal state of Brandenburg in north-east Germany, only around 3,000ha are still mires and another 4,000 ha are under restoration (Luthardt 2014a; LfU 2016). A primary focus for rewetting projects is the re-establishment of mire biodiversity. Therefore, its assessment is of great importance for gauging the success of restoration (cf. Luthardt & Wichmann 2016).

Typical biodiversity assessments based upon species richness - such as the concept of alpha, beta and gamma biodiversity (Whittaker 1972) - are not suitable for mires (Littlewood et al. 2010). A broader approach is needed, encompassing heterogeneity at different levels and a variety of functional elements (Minayeva et al. 2017). In order to evaluate the biodiversity of mires, their naturalness as well as their natural ecosystem functions and processes must be considered (Bragg & Lindsay 2003; Prentice 2011). This is shown in recent multifunctional restoration assessments focussing not on single taxa but on plant diversity, water table, peat decomposition, water holding capacity, and nutrient level (Strobel et al. 2019). The unique characteristics of each mire biodiversity component should be key elements for its assessment and all components of mire biodiversity including specialist species, habitat conditions and morphological heterogeneity must be examined (Bragg & Lindsay 2003; Prentice 2011; Minayeva et al. 2017).

Genetic diversity is regarded as the basis of biodiversity because it enables the adaptation of species to selective pressures (Laikre et al. 2016). It determines important factors such as extinction risk, resilience to environmental change and the fitness of populations or individuals, but is influenced by habitat size and quality and can, therefore, be threatened by the fragmentation and isolation of habitats (Struebig et al. 2011; Crawford & Keyghobadi 2018). In the context of peatlands, there have been some investigations on the genetic variation of e.g. Sphagnum species (Stenøien & Flatberg 2000; Shaw et al. 2008; Yousefi et al. 2019), species interactions in bog-plant communities (Schwarzer et al. 2013), and the implications of conservation and management strategies for genetic diversity (Crawford & Keyghobadi 2018; Eschenbrenner et al. 2019). It is estimated that in future, the role of genetics will gain more importance and recognition in conservation practice due to the currently intensive research effort and the increasing number of available genome sequences for species (Allendorf et al. 2013).

Plant species play an important role in the function, characterisation and assessment of habitats (Kaiser et al. 2002). They are used in (biodiversity) monitoring programmes because they are sessile, provide an indication of their abiotic environment (e.g. Ellenberg et al. 2010), occur in a wide variety of ecosystems, exhibit fairly low seasonality and dependency on weather conditions, and the availability of field botanists with expertise in vascular plants is generally good (cf. Brunbjerg et al. 2018). More than other ecosystems, mires are characterised by highly specialised plant species that are adapted to water-saturated, often extreme pH and nutrient conditions; and are mostly very rare, endangered and declining (Minayeva et al. 2008; Littlewood et al. 2010; Prentice 2011; Aapala et al. 2014). Following drainage, the specialised species disappear in favour of more numerous but ubiquitous species (Luthardt & Wichmann 2016). Various publications (inter alia Landgraf 2007; Penttinen et al. 2014; Joosten et al. 2015) advise that vascular plants and mosses should be considered in any assessment or monitoring of the biodiversity of peatlands.

In general, animal taxa are more demanding of habitat properties than plants, and indicate environmental changes more promptly (Görn & Fischer 2011; Lehmitz et al. 2020). Because it is generally impossible to record all taxa within one biocoenosis, surrogate taxa that allow broad assumptions about the peatland's status are commonly chosen. A surrogate taxon should include a broad range of typical peatland species with wellknown ecology, that express sensitivity to habitat changes as changes in abundance (Görn 2016). Multi-taxon approaches for ecological assessment of peatlands which have been researched and/or applied have considered birds, butterflies, orthoptera, ground beetles, dragonflies, ants, oribatid mites and spiders in different combinations (e.g. Görn & Fischer 2011; Penttinen et al. 2014; Joosten et al. 2015; Tiemeyer et al. 2015; Lehmitz et al. 2020).

Spatial heterogeneity and morphological variation positively affect biodiversity by influencing the occurrence and distribution of species (Dauber et al. 2003; Walz 2011). They can enhance connectivity, the potential for niche formation and the availability of habitats (Ludwig 1991; Wulf 2001; Fontaine et al. 2007; Walz 2011). Mires show a wide range of physiognomy (morphological forms), and structural heterogeneity at different scales (biogeographic zone, mire massif, complex of phytocoenoses/microtope, phytocoenosis/microform, microcoenosis) is an essential part of mire biodiversity (Bragg

& Lindsay 2003; Minayeva et al. 2017). The juxtaposition of typical microhabitats such as hollows, lawns and hummocks, offering conditions ranging from wet to dry, favours the development of vegetation patterns and plant species diversity at small scale (Luthardt 2014b; Korpela et al. 2020). Habitat heterogeneity has also been identified as a key determinant of faunistic diversity in bogs (Krieger et al. 2019).

Connectivity between habitats can positively influence genetic exchange and, therefore, the potential of a population for adaptation; whereas isolation can increase the extinction risk (e.g. Herrmann et al. 2013; De Vriendt et al. 2016). Mires maintain relatively stable conditions of, for example, microclimate and water availability, and thus play an important role in the connectivity between ecosystems by offering refuge to species (Minayeva & Sirin 2012). It has been shown that the sizes and connectivity of peatland ecosystems are important for the long-term abundance of butterflies, due to possible genetic exchange as well as the effectiveness of protection from natural enemies and/or local weather extremes (Settele & Reinhardt 1999).

Ecosystem ecology links the biota to their physical surroundings, describing the integrated system of interactions between organisms and their environment. The essential biota of a terrestrial ecosystem are its animals, plants and decomposers; whereas the abiotic components are soil, water and atmosphere (Chapin et al. 2011). Although numerous regional types of mire ecosystems can be distinguished on the basis of differences in hydrology, ecology, geomorphology or genesis (Joosten et al. 2017a), all mires share many ecological functions and features due to similar ecohydrological processes which are based on permanent water saturation and the accumulation of organic matter as peat (Parish et al. 2008). In mires, the water table is maintained at a level close to the ground surface by groundwater, surface water inflow or an excess of precipitation over evapotranspiration resulting in a positive climatic water balance (Edom 2001; Parish et al. 2008). Peat accumulates naturally under water-saturated conditions. When a mire is drained, peat formation is replaced by secondary pedogenetic processes (mineralisation, humification, shrinkage, consolidation, compaction, dislocation, leaching and accumulation of soil substances), leading to a hydrophobic topsoil with reduced water regulation and storage functions (Stegmann & Zeitz 2001; Zeitz

2016). Therefore, water table depth (below ground surface) and topsoil condition are used in peatland biodiversity assessments as indicators of the overall state of the ecosystem (Landgraf 2007; Klingenfuß et al. 2015).

As the assessment and monitoring of peatlands is often focussed only on parts of biodiversity components (mainly vegetation) and is often based on case studies (inter alia Duinen et al. 2002; Mälson et al. 2008; González et al. 2014; Renou-Wilson et al. 2019), tools that allow comparison and specifically address mire-specific biodiversity, are missing. In this article, we describe the development of a method for assessment of mire-specific biodiversity, aiming to provide a system that is practitioner friendly and transferable between geographical regions. Assuming that the complex composition of biodiversity cannot be assessed effectively on the basis of a single indicator (Hill et al. 2016), we adopted a multi-indicator approach that allows mire-specific biodiversity to be rated in terms of its heterogeneity at different levels. To calibrate the system we sampled 30 peatlands in the federal state of Brandenburg in north-east Germany, and carried out a validation exercise in collaboration with peatland experts and practitioners.

2.5. Methods

2.5.1. Basic criteria

We set the following five basic criteria for the eventual assessment method:

1. Different levels of mire-specific biodiversity shall be considered in the assessment. A single surrogate (e.g. vegetation) cannot be used successfully to assess the different levels of mire biodiversity, due to the different time delays in reactions to change of individual mire-specific factors and the spatial and morphological heterogeneity of mire-specific biodiversity at different levels and scales.
2. The focus will be on mire-specific characteristics. Thus, for example, the number of mire-specific species will be evaluated, rather than the total number of species.
3. All states of peatlands shall be represented, ranging from natural/unused to highly degraded/used as well as restored.

4. To ensure usefulness of the assessment method in practice, only features whose mire-specific attributes or components have been well researched and are accessible for the focus region shall be considered.
5. The final indicator system shall be practitioner friendly and cost effective.

2.5.2. Literature screening

We identified the relevant components of mire-specific biodiversity according to literature (Bragg & Lindsay 2003; Landgraf 2007; Parish et al. 2008; Littlewood et al. 2010; Görn & Fischer 2011; Prentice 2011; Minayeva & Sirin 2012; Aapala et al. 2014; Penttinen et al. 2014; Joosten et al. 2015; Klingenfuß et al. 2015; Tiemeyer et al. 2015; Minayeva et al. 2017; see also Introduction). We defined 'mire-specific biodiversity' as all biodiversity components that are exclusively adapted to functioning mires; and 'mire-typical biodiversity' as all biodiversity components that are highly adapted to mires but also occur in degraded mires and other peat and non-peat-forming wetlands. For each component, we identified suitable measurable indicators. The integrated system of indicators was developed for our study area in north-east Germany, but we place special emphasis on describing the methods we applied in order to demonstrate and enable transferability to other regions.

2.5.3. Field data and scaling

During the period 2002 to 2020, members of Eberswalde University for Sustainable Development collected biodiversity data from peatlands in the German federal state of Brandenburg, which encompass a wide range of different hydroecological mire types arising from the effects of different glacial influences (cf. Succow & Jeschke 1986; Kühn 2014). Although peatlands in the state have been drained intensively, about 3000 ha of mires are present (Luthardt 2014a). These peatlands were classified from natural to non-natural following the method for defining hemeroby as the level of human influence of peatlands developed by Wagner & Wagner (2005) (Table 2.1). For each of the hemeroby classes the data for six peatlands belonging to diverse ecological mire types (acidic to calcareous and nutrient poor to nutrient rich) were analysed (see Figure 2.3 for locations of the 30 sites selected and Table 2.7 for mire types). Due to degradation processes resulting from drainage (particularly mineralisation and the release of nutrients), eutrophic peatlands were represented more frequently in the three 'anthropogenic' categories. Based on the analysed data, ordinal scales defining 'low' to 'high' mire-specific biodiversity for each attribute of the indicator system were defined. The scale values range from 0 (not mire-specific) to 5 (highly mire-specific) in each case. This aligns with the five stages of peatland naturalness described by Wagner & Wagner (2005), with addition of a 'stage 0' for no mire-specific biodiversity as described by Tiemeyer et al. (2015). To reach an overall assessment, the values for all indicators were summed and again classified from 'no mire-specific biodiversity' to 'high mire-specific biodiversity'

Table 2.1: Stages of naturalness and hemeroby (human modification) of peatlands (translated and slightly modified from Wagner & Wagner 2005).

	Stage of naturalness/ hemeroby	Level of human influence	Nutrient balance (input/output)	Hydrology (drainage)	Vegetation (indicator species)
Natural	Natural (ahemerob to oligohemerob)	None to very low	Undisturbed	Undisturbed	Undisturbed
	Near-natural (oligohemerob to mesohemerob)	Low	Slightly disturbed	Slightly disturbed	Slightly disturbed
Athropogenic	Culturally-accentuated (mesohemerob)	Moderate	Moderately disturbed	Moderately disturbed	Moderately disturbed
	Culturally-characterised (euhemerob)	High	Highly disturbed	Highly disturbed	Highly disturbed; species indicating wet conditions still present
	Non-natural (polyhemerob)	Very high	Very highly disturbed	Very highly disturbed	Very highly disturbed; species indicating wet conditions missing

2.5.4. Expert validation and final calibration

Eleven experts in peatland restoration and 42 practitioners with experience in peatland restoration were interviewed, either individually (experts) or in a group workshop (practitioners). During these meetings, data on vegetation (complete list of vascular plants and mosses, highlighting mirespecific ones), physiognomy (plant formations and mire-typical special habitats), dominant water table and soil conditions as well as aerial pictures and representative photos for four (practitioners) or five (experts) peatlands was presented for assessment of the individual indicators and overall mire-specific biodiversity. To be able to compare these assessments with the assessment of our indicator system, the practitioners and experts were asked to assess each peatland subjectively. Therefore, they estimated the value for each indicator and overall mire-specific biodiversity based on their knowledge and expertise with peatlands using the scale from 0 to 5. The assessments by experts and practitioners were then compared with the assessment of our indicator system, and the outcome was used to optimise the calibration of our scales for assessment of biodiversity value.

2.5.5. Practical example

The 'Großes Brennbruch' (Brandenburg, Germany) is a complex of mesotrophic-acidic terrestrialisation mire and eutrophic paludification mire, which has been drained for forestry for over 130 years although the most intense interventions were applied in the late 1970s. In 2005 and 2006, the drainage systems within the 'Großes Brennbruch' and its catchment were dismantled in order to rewet the peatland (Koch 2007). To demonstrate the potential usage of our indicator system as a monitoring tool, we applied it to data collected before and after restoration of this peatland.

2.6. Results

2.6.1. The indicator system

The indicator system for measuring mire-specific biodiversity consists of three indicators representing (a) species level, (b) biocoenosis level and (c) ecosystem level. Genetic diversity is not included due to insufficient genetic research in our study area to support a genetic component for the assessment tool. Each indicator consists of two sub-indicators, each with defined mire attributes and metrics representing the essential components of the biodiversity level. The attributes and values to be measured have been well researched within the study area and are highly mire-specific or, at minimum, mire-typical (Figure 2.1). The indicators are based upon the cited literature and explained in detail below.

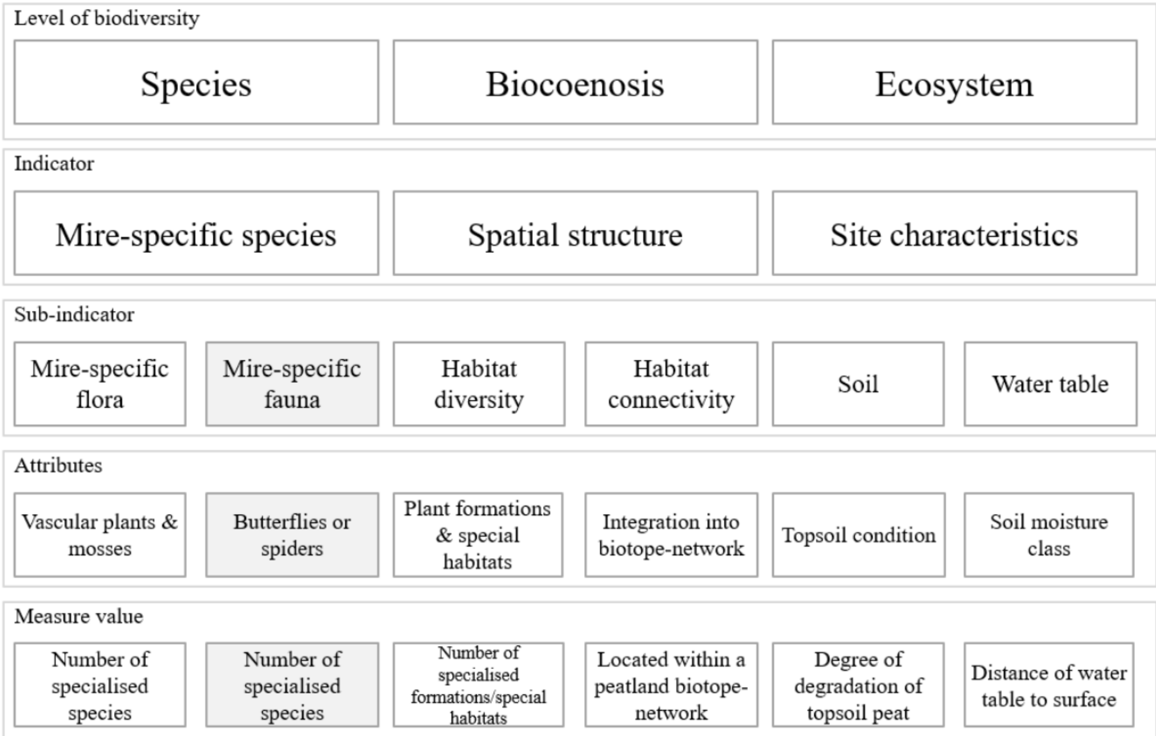


Figure 2.1: System of indicators for assessing mire-specific biodiversity. The attribute 'Butterflies or spiders' within the level of species was proposed as possibly available, but needed to be excluded within the suggested approach due to lack of data.

Species

The indicator for the species level is 'mire-specific species'. It consists of two sub-indicators: 'mirespecific flora' and 'mire-specific fauna'. For both sub-indicators, the value measured is the number of mire-specific species. 'Mire-specific flora' is described by the attribute 'vascular plants and mosses'. For our study area there is a complete revision of mire-typical and mire-specific vascular plants and mosses which lists 69 mire-specific vascular plants and 58 mire-specific mosses (Table 2.2; Klawitter 2014; Klawitter & Luthardt 2014; Luthardt 2014c). The species lists were derived from botanical and peatland literature (e.g. Zimmermann et al. 2004) and was checked by regional experts in botany and peatland science. To describe 'Mire-specific fauna' we only suggest the attribute ‘butterflies or spiders’. Data for butterflies and araneomorph spiders were not available for the sampled peatlands, so the subindicator ‘mire-specific fauna’ was excluded. There is no complete list of mire-specific and mire-typical fauna for our study region although some insights are provided by Luthardt & Zeitz (2014) who describe specialisation of mammals (Dolch 2014), birds (Flade 2014), amphibians (Brauner 2014a), butterflies (Gelbrecht 2014), dragonflies (Mauersberger 2014) and locusts (Brauner 2014b) as well as ground beetles, cicada, bugs, web spiders, pseudoscorpions and millipedes (Barndt 2014). Within this research, butterflies show a high specialisation on mires and peatlands and there are many practising lepidopterists (Gelbrecht 2014). Furthermore, araneomorph spiders show potential usability for a biodiversity assessment and 17 species are known to be mire-specific (Barndt 2014; Platen 1989).

Biocoenosis

The indicator for the biocoenosis level is the ‘spatial structure’ representing different plant sociologies and their associated fauna. It consists of two subindicators, ‘habitat diversity’ and ‘habitat connectivity’.

‘Habitat diversity’ is described by the attribute ‘plant formations and special habitats’. To assess the diversity of spatial structures at microtope level (Minayeva et al. 2017), we used the classification of physiognomic heterogeneity “Tentative physiognomic-ecological classification of plant formations of the earth”, which describes plant formations as “combinations of plant life forms, i.e. as physiognomic units” (Ellenberg & MüllerDombois 1966) and can be applied to describe different, adjoining formations in a single peatland area. Plant formations have the advantage of being globally recognisable and can easily be defined and described at regional level. By reviewing the zonation of vegetation on natural mires in our study area, we derived 15 regionally occurring mire-specific and mire-typical plant formations (Table 2.3). To represent a smaller-scale physiognomic level we also defined special habitats (based on Luthardt 2014b, Minayeva et al. 2017) that are mire-specific or mire-typical and occur in natural mire ecosystems

(Table 2.4). The attribute 'habitat diversity' is evaluated as the number of different specialised plant formations and special habitats.

Table 2.2: Mire-specific vascular plant and moss species occurring in the federal state of Brandenburg (north-east Germany).

Mire-specific vascular plant species (Luthardt 2014c)	Mire-specific moss species (Klawitter & Luthardt 2014)
Andromeda polifolia, Betula humilis, Betula nana, Betula pubescens, Blysmus compressus, Calla palustris, Carex (C.) appropinquata, C. cespitosa, C. chordorrhiza, C. davalliana, C. diandra, C. dioica, C. echinata, C. elata, C. flacca, C. flava, C. lasiocarpa, C. lepidocarpa, C. limosa, C. panicea, C. paniculata, C. pulicaris, C. rostrata, C. vesicaria, Cladium mariscus, Comarum palustre, Drosera intermedia, Drosera longifolia, Drosera x obovata, Drosera rotundifolia, Eleocharis mamillata, Eleocharis multicaulis, Eleocharis quinqueflora, Epipactis palustris, Eriophorum (E.) angustifolium, E. gracile, E. latifolium, E. vaginatum, Gentianella uliginosa, Hammarbya paludosa, Hottonia palustris, Juncus alpinus, Juncus filiformis, Juncus subnodulosus, Ledum palustre, Liparis loeselii, Lycopodiella inundata, Menyanthes trifoliata, Myrica gale, Parnassia palustris, Pedicularis palustris, Pedicularis sylvatica, Rhynchospora alba, Rhynchospora fusca, Saxifraga hirculus, Scheuchzeria palustris, Schoenus ferrugineus, Schoenus nigricans, Stellaria crassifolia, Trichophorum alpinum, Trichophorum cespitosum, Triglochin palustre, Utricularia australis, Utricularia intermedia, Utricularia minor, Utricularia stygia, Vaccinium macrocarpon, Vaccinium oxycoccus, Viola epipsila	*Bryum longisetum, Calliergon stramineum, Calliergon trifarium, Calypogeia sphagnicola, Cephalozia connivens, Cephalozia macrostachya, Cephalozia pleniceps, Cephaloziella elachista, Cephaloziella spinigera, Cladopodiella fluitans, Dicranum bergeri, Drepanocladus cossonii, Drepanocladus lycopodioides, Drepanocladus revolvens, Fissidens osmundoides, Hamatocaulis vernicosus, Helodium blandowii, Leiocolea rutheana, Lophozia laxa, Meesia hexasticha, Meesia longiseta, Meesia triquetra, Meesia uliginosa, Mylia anomala, Paludella squarrosa, Pohlia sphagnicola, Polytrichum commune, Polytrichum strictum, Scapania paludicola, Sphagnum (S.) affine, S. angustifolium, S. balticum, S. capillifolium, S. centrale, Sphagnum compactum, S. contortum, S. cuspidatum, S. denticulatum var. denticulatum, S. denticulatum var. inundatum, S. fallax, S. fimbriatum, Sphagnum flexuosum, S. fuscum, S. magellanicum, S. majus, Sphagnum molle, S. obtusum, S. papillosum, S. platyphyllum, S. riparium, S. rubellum, S. subsecundum, S. tenellum, S. teres, S. warnstorfii, Splachnum ampullaceum, Tomentypnum nitens, Warnstorfia fluitans*

Table 2.3: Mire-specific and mire-typical plant formations occurring in the federal state of Brandenburg (northeast Germany). Remark: Species inventory and micro-relief as described in Ellenberg & Müller-Dombois 1966, but hydromorphologically no bogs are present in Brandenburg).

Formations class	Formation subclass	Formation group	Formation
Closed forests	Mainly deciduous forests	Cold-dedicious forests without evergreen leaves	Cold-deciduous swamp or peat forest
Fourrés (shrubs)	Mainly deciduous fourrés (shrubs)	Cold deciduous shrublands (or thickets)	Deciduous peat shrubland (or thicket)
Dwarf-shrubs and related communities	Mossy bog formations with dwarf-shrubs	Raised bogs	Subcontinental woodland bog
		Non-raised bog	Blanket bog
Terrestrial herbaceous communities	Sedge swamps and flushes	Sedge peat swamps and similar swamps	Tall sedge swamp with creeping sedges
			Tall sedge swamp with caespitose sedges
			Low sedge swamp
		Flushes	Forb flushes (calcareous)
			Forb flushes (non calcareous)
			Moss flush (calceareous)
			Moss flush (non calcareous)
Aquatic plant formations	Floating meadows	Mainly herbaceous floating meadows	Temperate and subpolar herbaceous floating meadows
		Mainly mossy floating meadows	Mossy floating meadow
	Reed swamps	Reed-swamp formations of fresh water lakes	Temperate and subpolar fresh water reed-swamps
		Reed-swamp formations of flowing water	Temperate reed swamps on river banks

Table 2.4: Mire-specific and mire-typical special habitats occurring in the federal state of Brandenburg (northeast Germany).

Mire-specific special habitats	Mire-typical special habitats
Hummock, hollow, lagg (“fen strip separating a bog from the surrounding mineral soil” (Joosten *et al.* 2017), running spring water	Lying and upright dead wood, root plates, open water body (temporary or permanent), mineral islands, solitary trees, areas with no vegetation (e.g. mudbanks)

‘Habitat connectivity’ is described by the attribute ‘integration into biotope network’. The biotope networks described for the federal state of Brandenburg (Herrmann et al. 2013) identify core and connecting areas within which species exchange is possible. They include two well-connected networks involving peatlands, namely ‘small mires and peatlands in forests’ and ‘wet pastures and fens of the glacial valley’ (Figure 2.2). These were derived by defining criteria (size, protected areas, wet biotopes) for delineation of the core zones, then adding 1000 m buffer areas to form the networks. An indicator value (1) is recorded for this attribute if the peatland under examination is located within one network, using the network ‘small mires and peatlands in forests’ for peatlands located in forests and the network ‘wet pastures and fens of the glacial valley’ for peatlands located in the glacial valley and/or in agriculturally characterized surroundings.

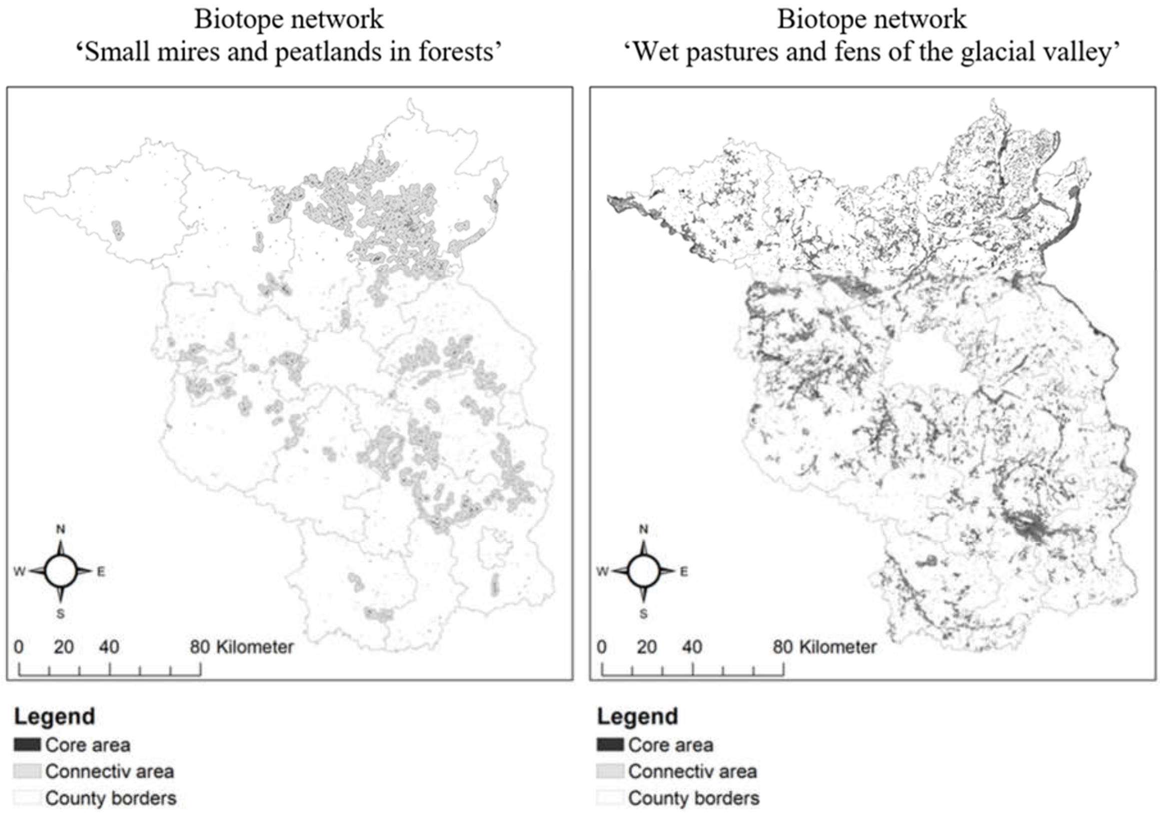


Figure 2.2: Biotope network of peatlands in Brandenburg, north-east Germany (Herrmann et al. 2013).

Ecosystem

The indicator for this level refers to the abiotic components of the ecosystem, or ‘site characteristics’. It consists of two sub-indicators, namely ‘soil’ and ‘water table’.

‘Soil’ is described by the attribute ‘topsoil condition’ and is measured in terms of degree of degradation of the first upper homogenous horizon of the soil profile. To assess peat degradation, different development stages of the topsoil ranging from peat accumulation (undegraded peat) to murshified peat (highly degraded peat) are defined, based on Schulz et al. (2019) (Table 2.5).

Table 2.5: Stages of peat topsoil (uppermost 30 cm) degradation from non-degraded peat (currently forming) to no peat.

Stage	Description
Non-degraded peat (currently forming)	Peat of low decomposition (‘fibric’ (Joosten et al. 2017)) in nutrient poor, acidic, base-rich or calcareous mires (e.g. moss peats, herbaceous peats with radicels and rhizomes); in naturally eutrophic mire ecosystems with or without natural water level fluctuations, such as alder forests, also moderately decomposed (‘hemic’) peat can occur (e.g. herbaceous peats with radicels and rhizomes or wood peat) (Schulz et al. 2019).
Non-degraded peat (currently not forming) or gyttja	Dry peat of low to moderate decomposition (‘hemic’ (Joosten et al. 2017)) (divisions as above) as well as gyttja, meaning a sedentarily accumulated material that consist of at least 5 % (dry mass) of organic matter (Schulz et al. 2019).
Slightly degraded peat (highly decomposed peat)	Highly decomposed peat (‘sapric’ (Joosten et al. 2017)); ‘Compact, mainly homogeneous, dark brown to black mass; unstructured (amorphous) or aggregated into larger pieces; muddy to mushy consistency when wet, comparable to a squeezed-dry sponge when dry; no or a small amount of recognisable plant remains; plant remains usually limited to more highly decomposed wood or fibre fragments’ (Schulz et al. 2019).
Moderately degraded peat (earthified peat)	‘Dark brown to black-brown mass with crumb grain structure, consisting of bonded soil particles of various sizes (but mainly >1 mm); similar to garden mould; smeary consistency when wet, crumbly but never powdery-dusty when dry; no or only a small amount of recognisable plant remains’ (Schulz et al. 2019).
Highly degraded peat (murshified peat)	‘Black-brown to deep black, loose mass with fine granular structure, consisting of small (mainly <1 mm) bonded soil particles; thick, silty mass when very wet, smeary-granular when moist, distinctly granular and powdery-dusty when dry (resembling loose coal slack); no recognisable plant remains’ (Schulz et al. 2019).
No peat	All soil substrates that are not peat (defined in Germany as sedentarily accumulated material that consists of more than 30 % (dry mass) of incompletely decomposed plant remains and humic substances or gyttja (Schulz et al. 2019).

‘Water table’ is described by the ‘soil moisture class’. Koska (2001) developed the concept of ‘vegetation forms’ for peatlands and wetlands in north-eastern Germany, which employs vegetation as a proxy for water table relative to the ground surface. It is thus possible to determine areas with different water tables, which are described by long-term median values of (positive or negative) standing water depth during wet and dry seasons (Table 2.6). The soil moisture class 5+ is most favourable for peat formation (cf. Parish et al. 2008; Joosten et al. 2015). The attribute ‘soil moisture class’ is evaluated as the distance from the water table to the soil surface, which is close to zero in natural mires.

Table 2.6: Soil moisture classes and associated water tables for peatlands (Joosten et al. 2015 after Koska 2001). NV = no value.

Soil moisture class	Verbal description	Water table relative to ground surface (+ above, - below)		
		Long-term median water table in the wet season	Long-term median water table in the dry season	Water supply deficit
6+	Lower eulitoral	+150 to +10	+140 to +0 cm	No value
5+	Wet	+10 to -5 cm	+0 to -10 cm	No value
4+	Very moist	-5 to -15 cm	-10 to -20 cm	No value
3+	Moist	-15 to -35 cm	-20 to -45 cm	No value
2+	Moderately moist	-35 to -70 cm	-45 to -85 cm	No value
2-	Moderately dry	No value	No value	<60 l/m^2

In order to determine the dominant soil moisture class as well as the dominant degree of topsoil degradation, each site is subdivided into homogenous vegetation units (if there is more than one) firstly. Therefore, areas with homogenous floristic dominances and physiognomic structure are segregated from each other. All units are outlined on recent satellite images and transferred into a geographic information system to create spatial maps of each site. For each vegetation unit all plant species and their cover are recorded. To transfer these data into soil moisture classes, the water table indication of each plant species described by Koska (2001) is applied to determine the soil moisture class of each vegetation unit. Further, for each vegetation unit the first upper homogenous horizon of topsoil peat is estimated in the field and classified into the different stages (Table 2.5). Thereby, the dominant topsoil degradation can be spatially described, too.

2.6.2. The ordinal scales for mire-specific biodiversity

The data from the 30 sampled peatlands is summarised in Table 2.7. In the sub-sections that follow, we explain the scoring system for each level of biodiversity in turn, and then the derivation of the overall site score for mire-specific biodiversity. The resulting survey sheet for field assessment of mirespecific biodiversity is provided in Chapter 2.10 Appendix.

Species

Table 2.7 shows that the numbers of mire-specific vascular plants recorded for the 30 sampled peatlands range from a median value of 12 in natural peatlands to almost zero in non-natural peatlands; whereas mire-specific mosses seem to occur in natural to culturally-accentuated peatlands but hardly at all in culturally characterised and non-natural peatlands. On the other hand, the number of mire-typical vascular plant species is higher in degraded peatlands then in natural and near-natural sites. On this basis, we excluded mire-typical plant species from the assessment.

The derivation of ordinal values for the 'mirespecific species' indicator is shown in Table 2.8. The final value is based mostly on the number of mirespecific vascular plants, with a single point added if mire-specific mosses are also present. In other words, it is the overall presence, rather than the number, of moss species that is rated; and if mosses are not determined and cannot be accounted for, the effect on the overall evaluation is not severe. This approach was adopted because mosses are not typically identified to species level in common practice, on account of the need for expert knowledge.

Table 2.7: Mire type, mire-specific flora (total number and number of mire-typical and mire-specific vascular plants and mosses), spatial structure (number of mire-typical plant formations and special habitats, integration into biotope network), topsoil state (no peat to non-degraded peat in % of total area) and soil moisture class (2- to 6+ in % of total area) for the 30 sampled peatlands in Brandenburg (north-east Germany), classified according to stages of naturalness (Wagner & Wagner 2005).

			Species					Biocoenosis			Ecosystem											
			Mire-specific species					Spatial structure			Site characteristics											
Natu-ralness	Study site	Mire type	Mire-specific flora								Topsoil state (in % of total peatland area)						Soil moisture class (in % of total peatland area)*					
			A	**B**	**C**	**D**	**E**	**I**	**II**	**III**	**5**	**4**	**3**	**2**	**1**	**0**	**6+**	**5+**	**4+**	**3+**	**2+**	**2-**
natural	1	a	47	9	11	0	4	4	5	1	100	0	0	0	0	0	0	100	0	0	0	0
	2	a	64	21	13	2	3	5	7	1	85	15	0	0	0	0	12	88	0	0	0	0
	3	a	31	17	6	0	1	3	5	1	54	46	0	0	0	0	0	100	0	0	0	0
	4	b	102	43	19	11	7	7	5	1	nv	nv	nv	nv	nv	nv	20	60	20	0	0	0
	5	a,c	64	31	15	1	4	5	5	1	55	45	0	0	0	0	45	55	0	0	0	0
	6	a	48	16	11	1	2	3	5	1	50	35	15	0	0	0	0	100	0	0	0	0
	Mdn		**56**	**19**	**12**	**1**	**4**	**5**	**5**	**1**	**55**	**25**	**0**	**0**	**0**	**0**	**6**	**94**	**0**	**0**	**0**	**0**
near-natural	7	c,b	73	38	8	2	1	3	4	1	0	0	47	53	0	0	0	72	28	0	0	0
	8	a	27	4	8	1	4	2	3	1	8	2	90	0	0	0	0	8	2	90	0	0
	9	c	38	18	3	0	0	4	3	1	0	93	7	0	0	0	0	93	7	0	0	0
	10	c,a	66	26	7	0	1	5	7	1	0	8	66	24	2	0	0	22	58	20	0	0
	11	a	48	23	12	1	2	4	2	1	0	60	40	0	0	0	0	60	0	40	0	0
	12	c	42	19	4	0	1	1	3	1	0	18	12	8	62	0	0	38	62	0	0	0
	Mdn		**48**	**21**	**8**	**0.5**	**1**	**4**	**3**	**1**	**0**	**13**	**44**	**4**	**0**	**0**	**0**	**49**	**18**	**10**	**0**	**0**
culturally-accentuated	13	c	99	42	5	2	1	2	6	1	0	31	12	0	0	57	0	31	12	57	0	0
	14	b,c	47	30	2	0	0	4	3	1	nv	nv	nv	nv	nv	nv	0	100	0	0	0	0
	15	c,b	97	46	2	1	0	3	4	1	0	0	0	0	99	1	0	0	36	63	0	1
	16	c,a	31	10	5	0	1	2	4	1	15	20	55	0	0	10	0	15	15	45	25	0
	17	c	143	46	4	1	0	3	7	0	nv	nv	nv	nv	nv	nv	0	0	55	45	0	0
	18	a	29	10	5	0	1	2	3	1	0	87	13	0	0	0	0	13	0	87	0	0
	Mdn		**72**	**36**	**5**	**0.5**	**1**	**3**	**4**	**1**	**0**	**26**	**13**	**0**	**0**	**5.5**	**0**	**14**	**14**	**51**	**0**	**0**
culturally-characterized	19	c,a	67	17	6	3	3	2	7	0	2	0	0	74	24	0	0	2	24	63	11	0
	20	c	31	16	0	3	2	2	nv	1	nv	nv	nv	nv	nv	nv	0	0	32	68	0	0
	21	c	83	44	2	0	0	3	3	1	0	0	0	0	25	75	0	0	30	70	0	0
	22	c	106	41	1	0	0	3	7	1	0	1	3	89	5	2	0	2	28	60	0	0
	23	c	96	34	0	0	0	1	3	1	0	0	0	0	63	37	0	0	0	36	27	22
	24	c,b	60	27	4	1	0	2	7	1	10	0	26	0	48,3	16	0	10	26	64	0	0
	Mdn		**75**	**31**	**2**	**0.5**	**0**	**2**	**7**	**1**	**0**	**0**	**0**	**0**	**25**	**16**	**0**	**1**	**27**	**64**	**0**	**0**
non-natural	25	c	176	50	4	0	0	1	3	0	nv	nv	nv	nv	nv	nv	nv	nv	nv	nv	nv	nv
	26	c	81	26	0	0	0	3	2	1	0	0	0	0	46	54	0	0	33	66	1	0
	27	c	50	12	2	0	1	1	2	1	0	0	0	55	45	0	0	0	0	60	40	0
	28	c	21	5	0	0	0	0	2	1	0	17	37	46	0	0	0	0	0	83	17	0
	29	c	75	26	0	0	0	2	2	1	0	0	0	0	100	0	0	0	18	82	0	0
	30	c	96	27	1	0	0	0	0	1	0	0	0	0	80	20	0	0	19	61	0	20
	Mdn		**78**	**26**	**1**	**0**	**0**	**1**	**2**	**1**	**0**	**0**	**0**	**0**	**46**	**0**	**0**	**0**	**18**	**66**	**1**	**0**

Abbreviations:

a: Nutrient poor, acidic b: Nutrient poor, base-rich/calcareous c: Nutrient rich A:Total number of species of vascular plants & mosses B: Mire-typical vascular plants C: Mire-specific vascular plants D: Mire-typical mosses E: Mire-specific mosses	I: Number of plant formations II: Number of special habitats III: Integrated into biotope network (1=Part of network, 0=Not part network) Mdn: Median *definitions see Table 2.7 nv: No value	5: Non degraded peat (currently forming) 4: Non degraded peat (currently not forming) or gyttja 3: Slightly degraded peat (highly decomposed) 2. Moderately degraded peat (earthified peat) 1. Highly degraded peat (murshified peat) 0: No peat

Table 2.8: Value scale for indicator 'mire-specific species' based on number of mire-specific vascular plants and presence of mire-specific mosses (species diversity).

Number of mire-specific vascular plants	Value	Mire-specific mosses present?	Total value 'mire-specific species'
≥ 10	4	Yes	5
		No	4
≥ 7	3	Yes	4
		No	3
≥ 4	2	Yes	3
		No	2
≥ 1	1	Yes	2
		No	1
0	0	Yes	1
		No	0

Biocoenosis

Amongst the 30 sampled peatlands, the number of mire-specific and mire-typical plant formations increases with increasing naturalness (Table 2.7). Most of the non-natural and culturally-characterised peatlands have only one or two mire-specific and mire-typical formations which are mainly dominated by reeds or sedges. In contrast, the formations in natural and near-natural peatlands are divers, often comprising a combination of peat forests with sedge, reed and moss dominated formations. The number of special habitats per peatland fluctuates widely, but also generally increases with increasing naturalness. Laggs, hummocks, hollows and open water bodies are mostly recorded in the less-degraded peatlands. Almost all of the sampled peatlands are located within one of the peatland biotope networks (Figure 2.2).

The calculation of ordinal values for the 'spatial structure' indicator is illustrated in Table 2.9. Scores up to 3 are awarded on the basis of 'number of mire-specific and mire-typical plant formations', then one point is added if the number of special habitats is at least 3, and another if the peatland lies within a biotope network.

Table 2.9: Value scale for the indicator 'spatial structure' based on number of mire-specific and mire-typical plant formations, number of special habitats and integration into biotope network (biocoenosis diversity).

Number of mire specific & mire-typical plant formations	Value	Number of special habitats	Added value	Located within a biotope network	Total added value 'spatial structure"
≥ 5	3	≥ 3	4	Yes	5
				No	4
		0 - 2	3	Yes	4
				No	3
≥ 3	2	≥ 3	3	Yes	4
				No	3
		0 - 2	2	Yes	3
				No	2
≥ 1	1	≥ 3	2	Yes	3
				No	2
		0 - 2	1	Yes	2
				No	1
0	0	≥ 3	1	Yes	2
				No	1
		0 - 2	0	Yes	1
				No	0

Ecosystem

Topsoil condition in the sampled peatlands shows a shift from highly degraded in non-natural peatlands to non-degraded with current peat formation in natural peatlands, while the soil moisture class ranges from 2+/3+ in non-natural peatlands to mainly 5+ in natural peatlands (Table 2.7).

The matrix of ordinal values for the 'site characteristics' indicator is shown in Table 2.10. Each peatland is scored on the basis of the spatially dominant (most extensive) topsoil condition and soil moisture classes observed. Although open water (soil moisture class 6+) was very seldom recorded in the sampled peatlands, the second highest ranking for 'water table' is awarded if the dominant soil moisture class is found to be 6+. Peatland restoration measures can lead to surface flooding if the peat is so degraded that it cannot absorb inflowing water, i.e. if the peatland has lost its 'surface oscillation' (Mooratmung) function and acts hydrophobic due to oxidisation (Zeitz 2014 & 2016); and similar scenarios may arise in less-disturbed peatlands as the incidence of drought conditions increases due to climate change. On this basis, we decided that areas of shallow open water should score four points in order to attach value to the presence of a high water table under such circumstances, even though gyttja rather than peat will form in this situation.

Because the water table is the driving factor that enables peat accumulation (Joosten 2008), it is of higher impact than the topsoil state (Table 2.10)

Table 2.10: Matrix of values for the 'site characteristics' indicator, based on soil moisture class and topsoil state (ecosystem diversity).

Topsoil state / Soil moisture class	Non-degraded peat (currently forming)	Non-degraded peat (currently not forming) or gyttja	Slightly degraded peat (highly decom-posed peat)	Moderately degraded peat (earthified peat	Highly degraded peat (murshfied peat)	No peat or gyttja
5+	5	5	4	4	3	3
4+ or 6+	4	4	4	3	3	2
3+	4	3	3	3	2	2
2+	3	3	2	2	2	1
2-	3	2	2	1	1	1
lower than 2-	2	2	1	1	0	0

Overall assessment of mire-specific biodiversity

After assessing the levels of biodiversity individually, the ordinal scores (0 to 5) are summed to give a cumulative score for the peatland and the cumulative scores are again classified from zero (no mire-specific biodiversity) to five (very high mirespecific biodiversity) in line with the five stages of naturalness by Wagner & Wagner (2005) (Table 2.11).

Using the scores as an overall description, the mire-specific biodiversity can be compared between two or more sites or, for a single peatland, before and after rewetting or across another defined time interval. Possible visualisations are shown in Figure 2.3 and, for the example of our 30 study sites, in Figure 2.4.

Table 2.11: Overall assessment of mire-specific biodiversity based on the accumulated indicator values for species, biocoenosis and ecosystem levels.

Class	Accumulated values	Verbal description	Colour code
5	14, 15	Very high mire-specific biodiversity	
4	11, 12, 13	High mire-specific biodiversity	
3	8, 9, 10	Moderate mire-specific biodiversity	
2	5, 6, 7	Low mire-specific biodiversity	
1	2, 3, 4	Very low mire-specific biodiversity	
0	0, 1	No mire-specific biodiversity	

2.6.3. Expert validation

The experts and practitioners who participated in the evaluation returned very similar assessments of mire-specific biodiversity, both overall and for the individual levels (Table 2.12). By comparing the assessment via the indicator system and the median of the evaluation by practitioners, a high conformity is visible. Eighty-one percent of the assessments were identical. Comparing the indicator assessment with expert assessment (median), 70 % of the assessments were identical. The similar outcomes of the mire-specific biodiversity assessments by the indicatorsystem and the practitioners and experts confirms the indication by our objectified system. So further adaptations were assessed as not necessary.

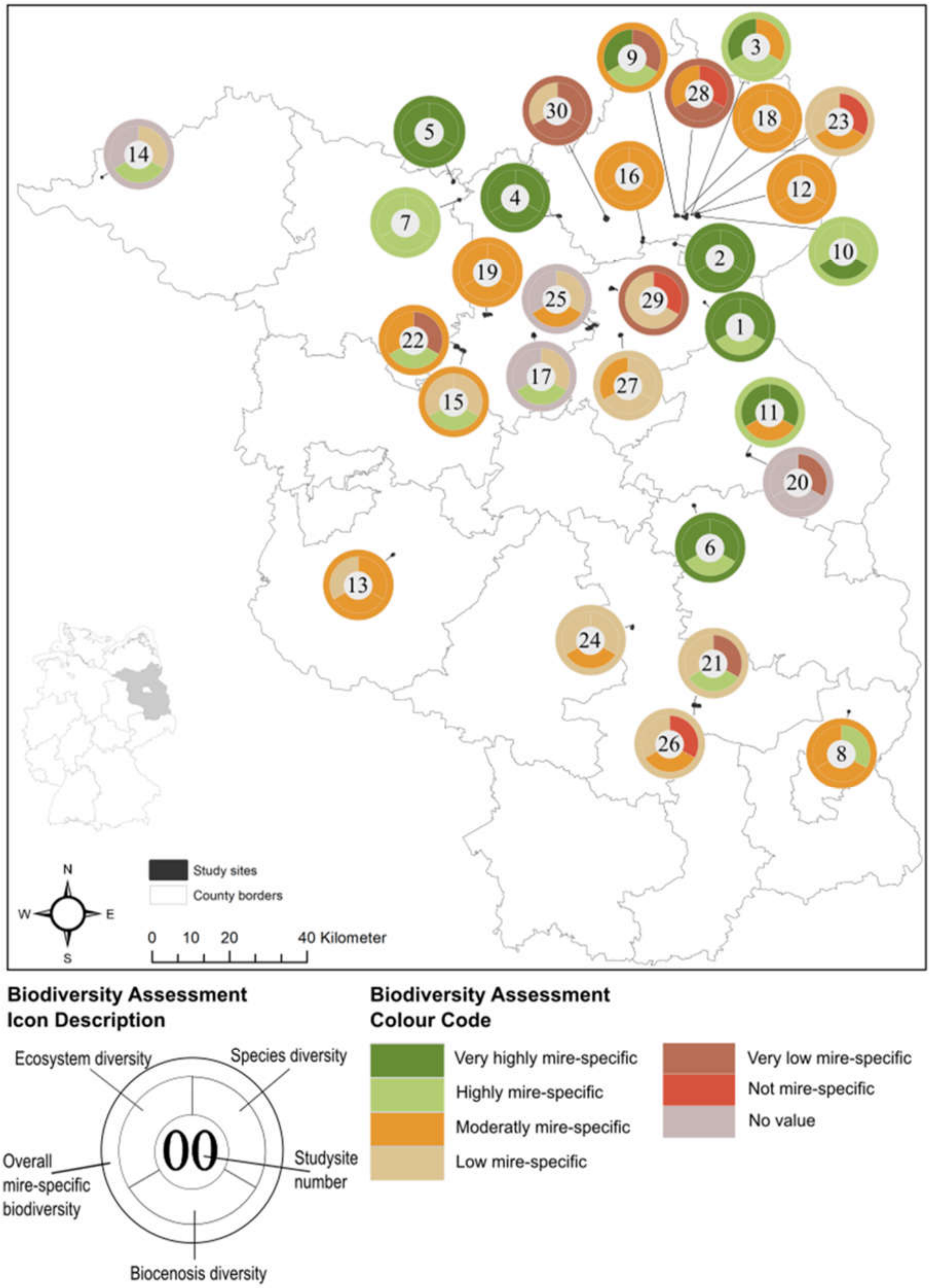


Figure 2.3: Visualisation of mire-specific biodiversity for the 30 sampled study sites in Brandenburg (northeast Germany (edited by J. Hammerich, source: County borders: VG250®ATKIS, ©BKG 2006).

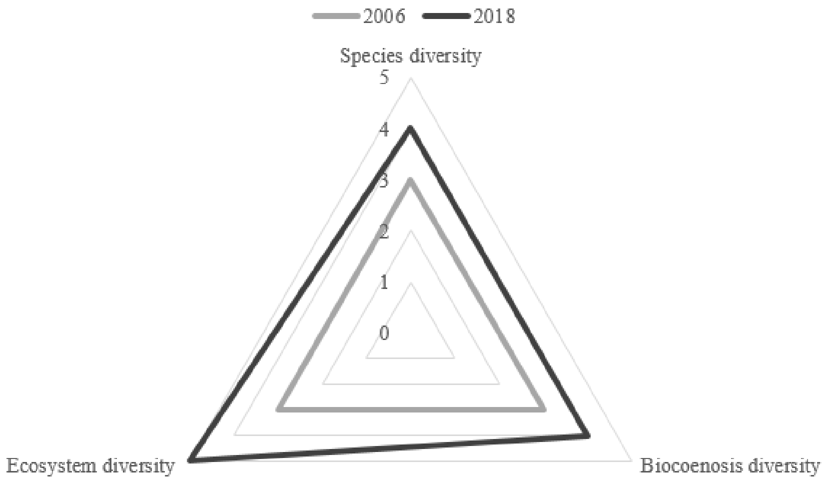


Figure 2.4: Visualisation of overall mire-specific biodiversity of "Großes Brennbruch" before (2006) and after (2018) restoration.

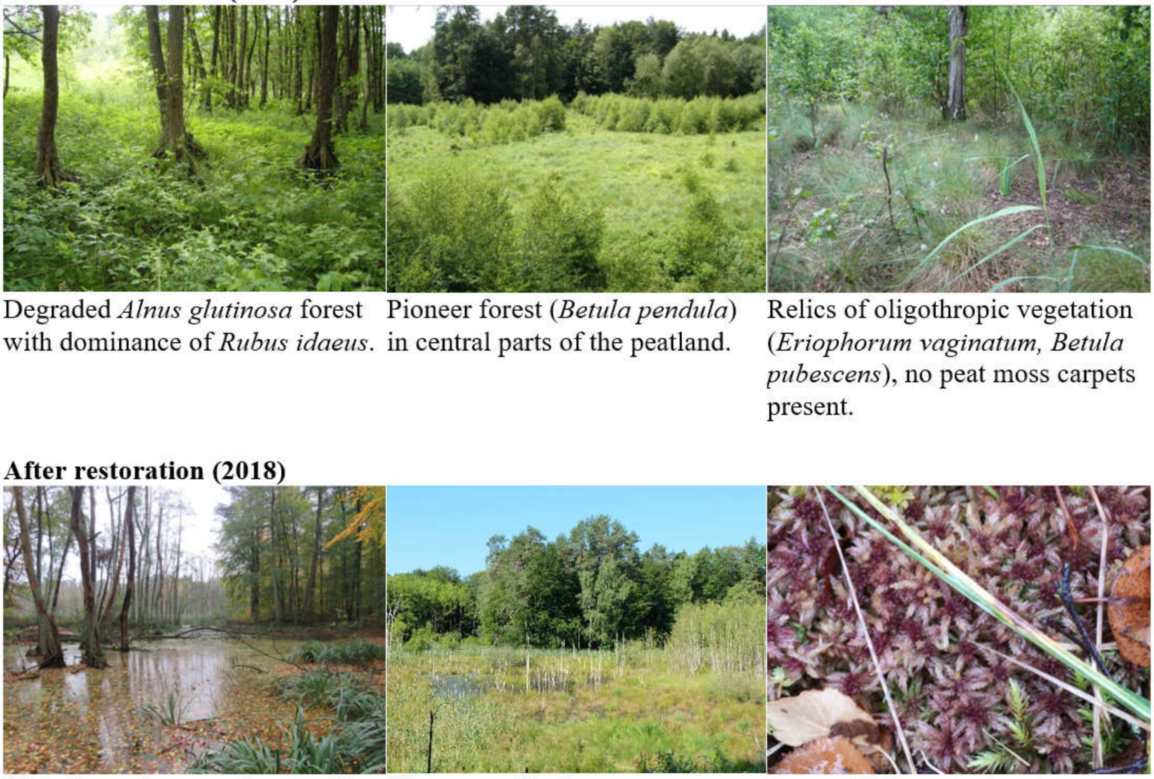
2.6.4. Case study

Before restoration in 2006, the topsoil at the 'Großes Brennbruch' mainly consisted of earthified peat and the dominant soil moisture class was 3+ (Table 2.13). Only the central *Sphagna-Betula pubescens*-peat forest (15 % of total area) had a higher water table and non-degraded peat profile. The other vegetation formations were mostly not mire-typical or specific. Areas dominated by *Calamagrostis epigejos* and *Rubus ideaus*, an alder forest with *Dryopteris cathusiana* in the herb layer and a *Betula pendula* pioneer forest were present (Figure 2.5). The only mirespecific formations were a cold -deciduous peat forest with *Betula pubescens* and *Sphagna*, and a temperate freshwater reedbed dominated by *Phragmites australis*. Five different mire-specific vascular plants were present, namely *Betula pubescens*, *Eriophorum vaginatum* and *Oxycoccus palustris* mainly in the central part of the peatland, but with *Carex elata* and *Carex paniculata* in the periphery.

Table 2.12: Comparison of assessment (species, biocenosis and ecosystem level as well as overall biodiversity) via indicator system and experts/practitioners.

Practitioner Workshop (n=42)																								
Study case	Species level						Biocoenosis level						Ecosystem level						Overall assessment					
	I	Practitioners					I	Practitioners					I	Practitioners					I	Practitioners				
		M	Mdn	Min	Max	Mf		M	Mdn	Min	Max	Mf		M	Mdn	Min	Max	Mf		M	Mdn	Min	Max	Mf
1	5	4.5	5.0	3	5	5	4	4.3	4.0	3	5	5	5	4.5	4.5	4	5	5	5	4.4	4.0	3	5	5
2	2	1.7	2.0	1	3	2	2	2.1	2.0	1	3	2	1	1.4	1.0	1	4	1	2	1.8	2.0	1	4	2
3	2	1.7	2.0	1	3	2	2	1.9	2.0	1	3	2	3	1.4	1.0	1	2	1	2	1.9	2.0	1	3	2
4	2	2.4	2.0	1	4	3	3	2.9	3.0	2	4	3	4	3.6	4.0	2	5	4	3	3.0	3.0	2	4	3
Expert Interviews (n=11)																								
Study case	Species level						Biocoenosis level						Ecosystem level						Overall assessment					
	I	Experts					I	Experts					I	Experts					I	Experts				
		M	Mdn	Min	Max	Mf		M	Mdn	Min	Max	Mf		M	Mdn	Min	Max	Mf		M	Mdn	Min	Max	Mf
5	4	3.4	3.0	2	4	4	4	4.0	4.0	3	5	4	5	4.7	5.0	4	5	5	4	3.9	4	3	5	3
6	1	0.7	1.0	0	2	0	3	2.4	2.0	1	4	2	2	2.1	2.0	1	5	2	2	1.7	1.5	1	3	1
7	5	4.5	5.0	3	5	5	5	4.4	4.0	3	5	5	5	4.4	5.0	2	5	5	5	4.8	5	4	5	5
8	1	1.4	1.0	0	3	1	3	3.0	3.0	2	4	3	3	2.2	2.0	1	4	2	2	2.2	2.0	1	4	2
9	1	1.2	1.0	0	4	1	3	3.0	3.0	0	5	4	4	4.0	4.0	2	5	5	3	3.0	3.0	1	4	3

Abbreviations:	Study cases:
I= Indicator based assessment	1 = 'Große Mooskute'
n = Total sample size	2 = 'Römerwiese'
M = Mean	3 = 'Eiserbuder Erlenwald'
Mdn = Median	4 = 'Kranichbruch'
Min = Minimum	5 = 'Großes Brennbruch'
Max = Maximum	6 = 'Koppainz'
Mf = Most frequent value	7 = 'Plötzendiebel'
	8 = 'Sernitz'
	9 = 'Rothsche Wiese'



Before restoration (2006)

Degraded *Alnus glutinosa* forest with dominance of *Rubus idaeus*.

Pioneer forest (*Betula pendula*) in central parts of the peatland.

Relics of oligothropic vegetation (*Eriophorum vaginatum, Betula pubescens*), no peat moss carpets present.

After restoration (2018)

Wet *Alnus glutinosa* forest with dying *Alnus glutinosa*.

Wet area with *Comarum palustre* and *Juncus effusus* in central parts of the peatland.

Re-establishment of peat moss carpets (*Sphagnum magellanicum)* and oligotrophic vascular plants.

Figure 2.5: Photo documentation 'Großes Brennbruch' 2006 (before restoration) and 2018 (after restoration).

In 2018, the topsoil showed reinstated peat formation, mainly by peat mosses over highly decomposed peat, and the dominant soil moisture class was 5+. In addition to the cold deciduous peat forest with *Betula pubescens* and peat mosses recorded in 2006, two other mire-typical formations were present, namely a reed swamp dominated by *Phragmitis australis* and a tall sedge swamp dominated by *Carex acutiformis*. The formerly dry alder forest was characterised by species indicating wetness (e.g. *Lemna minor, Hottonia palustris, Utricularia vulgaris*) and some sedges (e.g. *Carex acutiformis, Carex riparia*) (Figure 2.5). Special habitats (dead wood, open water pools, hummocks) could be found within all vegetation areas. Seven mire-specific vascular plants were present in nearly all vegetation areas, and peat mosses were growing on more than 50 % of the total area.

The overall rating for mire-specific biodiversity changed within 12 years from moderate (8/15 points) to high (13/15 points) (Figure 2.3).

Table 2.13: Overview of mire-specific biodiversity components at 'Großes Brennbruch' in the years 2006 (before restoration) and 2018 (after restoration).

Mire-specific vascular plants	Mire-specific mosses	Mire-specific plant formations	Special habitats	Located within peatland biotope network	Dominant topsoil condition	Dominant soil moisture class
2006						
Betula pubescens, Carex elata, Carex paniculata, Eriophorum vaginatum, Oxycoccus palustris	*Spagnum sp.*	cold-deciduous peat forest, temperate freshwater reed	hummocks, upright dead wood, mineral islands, areas with no vegetation	yes	Moderately degraded peat (earthified peat)	3+
2018						
Betula pubescens, Calla palustris, Carex elata, Carex lasiocarpa, Comarum palustre, Eriophorum vaginatum, Hottonia palustris	*Polytrichum commune, Sphagnum fallax, Sphagnum fimbriatum, Sphagnum magellanicum*	cold-deciduous peat forest, temperate freshwater reed, tall-sedge swamp	hummocks, hollows, lying dead wood, upright dead wood, open water body, mineral islands, areas with no vegetation	yes	Non-degraded peat (peat forming)	5+

2.7. Discussion

2.7.1. Indicators, attributes, measured values

We developed a multi-indicator assessment tool for mire-specific biodiversity, which considers the vegetation, habitat heterogeneity and connectivity, water table and topsoil degradation of the focus peatland. Existing suggestions on how to assess peatland biodiversity mainly focus on vegetation and fauna alone. Tiemeyer et al. (2015) suggest an approach where mire-typical vegetation is assessed by biotope value. Biotope values are a procedure that assigns a value to biotopes on the basis of their importance for nature conservation. This procedure is used in parts of Germany to compensate for interventions in nature (Deutscher Bundestag 2018). This value is then augmented by awarding ‘peatland points’ for natural, peat accumulating or peat preserving biotope types and their degraded states. The method for biodiversity of fauna is not fully developed, but the authors suggest an assessment based on the Red-List endangerment and the binding of species to mire-typical biotope types. In the context of integrating additional ecosystem services into carbon credits, Joosten et al. (2015) suggest two assessments for the biodiversity of mires - a costeffective standard approach and a premium approach. The first of these aims to compare the biotope value before and after restoration. The second is based on field surveys and suggests rating the abundance of vascular plants and mosses, amphibians, birds and arthropods. The model by Görn & Fischer (2011) is used for birds and arthropods, whereas no assessment model is developed for amphibians, vascular plants and mosses. The Görn & Fischer (2011) approach is based on evaluation of fens for nature conservancy purposes via faunistic indicators, rather than direct assessment of mire-typical or mire-specific biodiversity. Therefore, they suggested locusts, ground beetles, butterflies and birds as suitable taxa, listed all fen-typical species, chose criteria upon which they should be assessed, and developed value scales based upon distribution as well as national and international endangerment. Otherwise, where the term ‘biodiversity’ is used in various publications on peatlands, the authors often research and refer to diversity of vegetation and fauna only (e.g. Agus et al. 2019; Payne et al. 2018; Harrison & Rieley 2018; Renou-Wilson et al. 2019; Sundari et al. 2020).

In line with the literature (Bragg & Lindsay 2003; Prentice 2011; Minayeva et al. 2017) on recommendations for mire-typical and mire-specific biodiversity assessment, we do not think that individual taxa or a focus on vegetation and fauna alone are suitable indicators for the entirety of mire-specific biodiversity, even though studies show that, for example, vascular plants can function as strong indicators for overall biodiversity across environmental gradients (Brunbjerg et al. 2018). In peatlands, often long degradation processes due to drainage as well as peatland restoration lead to diverse states. For example, vascular plants and mosses often remain in retention areas or still-natural central areas even though the site is increasingly degrading overall. Taking additionally into account the dominant site conditions as well as structural heterogeneity gives a better understanding of the overall mire-specific biodiversity. In this way the maturity and functionality of the peatland ecosystem, which plays an important but often undervalued role for less-researched taxa such as ground beetles (Barndt 2014), are better taken into account.

The ecological conditions, in particular nutrient content and pH value, are commonly used in conservation projects or peatland description (Klingenfuß et al. 2015) to assess the peatland's state relative to the original or natural state (target state of restoration) of a specific mire type. Bragg & Lindsay (2003) also stress that the evaluation of peatland biodiversity needs to be based on assessment of the same mire type based on the criteria naturalness and diversity (representativeness and rarity). Although the measurement of nutrient content and pH is not directly included, these and other ecological traits are represented within the ecological amplitudes of mire-specific vascular plants and mosses. Furthermore, our indicator system targets a state of stable ecosystem functioning where peat is dominantly accumulating, the water table is predominantly at or above the ground surface, the mire-specific habitat structure is diverse, and mire-specific species are present. In this way, we do not target a specific eco-hydrological mire type, but rather we target the specific characteristics that all mire types share.

Each of the attributes chosen for inclusion in our assessment has been defined in terms of regional characteristics, offers good data availability, and is practically applicable.

Tiemeyer et al. (2015) and Görn & Fischer (2011) include endangerment (Red Lists) within their assessment, whereas we evaluate peatland attributes in terms of mire-specificity. We consider that integrating Red-List endangerment in an assessment of mire biodiversity is not constructive, at least in the case of our study region. Of the mire-specific vascular plants for Brandenburg, 62 % are listed as highly endangered, at risk of extinction or extinct and only 7 % are not listed at all (LUA 2006). If we were to focus on Red-List status, nearly all mire-specific species would be valued for endangerment, but so would all other (not mire-specific) Red-List-species - such as species adapted to degradation stages of mires or even dry ecosystems.

Biotope values, as suggested by Joosten et al. (2015) and Tiemeyer et al. (2015), are not widely developed and are based upon different characteristics in different regions, so they would not be applicable across all regions. Also, they do not aim to directly highlight mire-specific characteristics and could, therefore, be misleading.

We considered butterflies and araneomorph spiders for assessing peatland fauna, which show high specialisation on mires. Görn & Fischer (2011) suggest birds, butterflies, locusts and ground beetles covering a range of different spatial scales. Lehmitz et al. (2020) suggest the inclusion of vegetation, ground beetles, oribatid mites and araneomorph spiders in an ecological assessment of peatlands, finding good correlations concerning moisture and habitats for the last two. Dragonflies were considered, but were excluded because their habitat is not the mire itself but the open water bodies within mires (Mauersberger 2014). Batzer et al. (2016) evaluate the roles of terrestrial invertebrates in peatlands of Europe, Canada, USA, and China and their value for peatland biodiversity assessment and state that further research is greatly needed. In general the use of fauna in environmental assessments is complicated by the effort required for data acquisition due to their high mobility, transient or hidden lifestyles and their adaptation to multiple spatially separated and differently structured habitats, as well as the highly specialised knowledge required for determination of some species (Bastian & Schreiber 1999). The use in practitioner friendly assessments is questionable. Still, it is plausible that, with further research, additional taxa could be included in the indicator system.

The assessment of biotope connectivity is based on a single study from the year 2006. Therefore, it will only be possible to detect changes in biotope connectivity (for example after rewetting) if the biotope network is updated on a regular basis. The plant formations are described globally and can readily be adapted to individual regions by adding or removing mire-specific and mire-typical formations based on their regional manifestations. In the indicator system developed here, peatland size is addressed only indirectly through the dataset, which includes peatlands with total areas ranging from 1 to 51 ha. The number of species belonging to a taxonomic group generally increases with increasing area (species-area relationship; inter alia Preston 1962) whereas mire-specific species often show high specialisation but low demand on habitat size. To evaluate whether peatland size influences the number of mire-specific species within the same peatland hemeroby class, further research is needed.

2.7.2. Transferability

We placed special emphasis on explaining how the attributes and measured values were developed and which data they were based on, in order to create an example for adaptation to other regions. This form of assessment can be applied elsewhere, by defining the mire-specific components (species, plant formations, special habitats, biotope networks) of the region addressed and analysing data from degraded to natural peatlands occurring within that region. A precondition is the availability of reference systems, which are in a natural state.

To determine the degree of the peatlands naturalness we referred to Wagner & Wagner (2005). Joosten & Clarke (2002) define the naturalness of peatlands as 'the quality of not having been deliberately influenced by human beings'. Bragg & Lindsay (2003) refer to naturalness as either the 'full display of all expected components of natural diversity' or the "lack of evident human disturbance" and name vegetation and surface patterns as valuable components for the evaluation of a peatlands naturalness. Mendes et al. (2019) show an example on how to cluster peatlands in four classes of naturalness based on the level of human interference: disturbed, altered, conserved and wild.

We think the list of mire-specific plants and mosses provided by Klawitter (2014), Klawitter & Luthardt (2014) and Luthardt (2014c) is valid for north-east Germany and could be used for whole Germany with slight modifications. Concerning mire-typical and mire-specific species, a good basis for determination of mire-specific plants is provided by Joosten et al. (2017), who offer insights about the characteristic vascular plants and mosses of mires and peatlands in various European countries at different levels of detail (e.g. Risager et al. 2017 (Denmark); Krebs et al. 2017 (Georgia); Stefanut et al. 2017 (Romania)). Literature from the research fields of botany and environmental science offers methodologies for developing a list of peatland-typical or peatland-specific vascular plants for a focus area; for example, Sotek 2010 (Pomerania) and Anderson & Davis 1997 (Maine) researched distribution patterns and/or habitat conditions of peatland plants. Also, routine publications from nature conservation can be taken into account, such as the lists of characteristic species prepared for the European Union NATURA 2000 directive and descriptions of plant sociology or biotope types.

There are various publications on mire-typical and mire-specific fauna including ground beetles ((Holmes et al. 1993 (Wales); aquatic invertebrates (Horwitz 1997 (Australia); non-biting midges (Rosenberg et al. 1988 (Canada) and oribatid mites (Behan-Pelletier & Bissett 1994 (Canada); Wisdom et al. 2011 (Ireland)). As mentioned above, the major challenge will be listing all mire-specific species and setting value scales for the region under consideration.

Plant formations for each region can be derived from Ellenberg & Müller-Dombois (1966). Special habitats will need to be adjusted for the focus region. Examples of possible additions to reflect habitat heterogeneity in different regions are frailejones (Espleletia spp.) for the páramo region of South America (Cárdenas et al. 2018) and palsa formations for Norway (Moen et al. 2017).

Concerning biotope connectivity, previous research can be applied either to develop a regional biotope network or to implement data on existing networks. Available guidelines for developing a biotope network employ different approaches, for example spatial conservation prioritisation (Jalkanen et al. 2020) or the use of focal species (Bani et al. 2002). A good overview of international (transboundary) networks which include (but do not focus on) peatland habitats can be found in Bennett & Wit (2001). Established regional biotope networks often provide descriptions of the methodology employed in their development and can, therefore, be used not only directly but also as guidelines/models for new networks (e.g. Metropolregion Hamburg 2019).

The sub-indicators 'soil' and 'water table' show good transferability, due to the ubiquitous ecohydrological processes (based on permanent water saturation and the accumulation of organic matter as peat) of peatlands (Parish et al. 2008). In this study, we chose to work with soil moisture classes because that is a regionally accepted methodology with good spatial resolution. For other regions it might be practical to use water tables measured at gauging stations. In some locations it may be necessary to consider other influencing factors, such as permafrost.

We encourage scientists to apply our research as a model for other geographical regions. Assessing the biodiversity of peatlands on the basis of mire-specific characteristics highlights their importance for biodiversity in general, and provides tangible evidence to support conservation planning at regional to global scales.

2.8. Acknowledgements

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2.9. Author contributions

JH, CD, VL and FT designed the study. JH and CD did the literature review and set up the first draft of the indicator system. JH revised the indicator system and sampled the data. The data were collected by JH, CD, CS and other researchers at Eberswalde University for Sustainable Development. JH prepared the manuscript with the help of CS and VL. VL critically reviewed the study and contributed central ideas and discussion points. All authors contributed to the final version of the manuscript.

2.10. Appendix

Mire-specific biodiversity – Assessment sheet			
Mire-specific biodiversity is measured at three levels: species (Section 2), biocoenosis (Section 3) and ecosystem (Section 4.). For each level, attributes are evaluated and recorded on this form. At the end of each section is a point classification which will be summed up at the end of the evaluation sheet for an overall assessment of the mire-specific biodiversity of the peatland in focus.			
1. General information			
Site:		Latitude/Longitude:	
Date:		Size (ha):	
Editor/Editing organisation:			
Hydrological mire type:		□ known	□ presumed
Ecological mire type (current):			
Ecological mire type (former):		□ known	□ presumed

2. Species diversity		
2.1.Mire-specific vascular plants		
Instructions: Please mark all of the mire-specific vascular plants occurring within the peatland area.		
□ *Andromeda polifolia,*	□ *Betula humilis*	□ *Betula nana*
□ *Betula pubescens*	□ *Blysmus compressus*	□ *Calla palustris*
□ *Carex appropinquata*	□ *Carex cespitosa*	□ *Carex chordorrhiza*
□ *Carex davalliana*	□ *Carex diandra*	□ *Carex dioica*
□ *Carex echinata*	□ *Carex elata*	□ *Carex flacca*
□ *Carex flava*	□ *Carex lasiocarpa*	□ *Carex lepidocarpa*
□ *Carex limosa*	□ *Carex panicea*	□ *Carex paniculata*
□ *Carex pulicaris*	□ *Carex rostrata*	□ *Carex vesicaria*
□ *Cladium mariscus*	□ *Comarum palustre*	□ *Drosera intermedia*
□ *Drosera longifolia*	□ *Drosera x obovata*	□ *Drosera rotundifolia*
□ *Eleocharis mamillata*	□ *Eleocharis multicaulis*	□ *Eleocharis quinqueflora*
□ *Epipactis palustris*	□ *Eriophorum angustifolium*	□ *Eriophorum gracile*
□ *Eriophorum latifolium*	□ *Eriophorum vaginatum*	□ *Gentianella uliginosa*
□ *Hammarbya paludosa*	□ *Hottonia palustris*	□ *Juncus alpinus*
□ *Juncus filiformis*	□ *Juncus subnodulosus*	□ *Ledum palustre*
□ *Liparis loeselii*	□ *Lycopodiella inundata*	□ *Menyanthes trifoliata*
□ *Myrica gale*	□ *Parnassia palustris*	□ *Pedicularis palustris*
□ *Pedicularis sylvatica*	□ *Rhynchospora alba*	□ *Rhynchospora fusca*
□ *Saxifraga hirculus*	□ *Scheuchzeria palustris*	□ *Schoenus ferrugineus*
□ *Schoenus nigricans*	□ *Stellaria crassifolia*	□ *Trichophorum alpinum*
□ *Trichophorum cespitosum*	□ *Triglochin palustre*	□ *Utricularia australis*
□ *Utricularia intermedia*	□ *Utricularia minor*	□ *Utricularia stygia*
□ *Vaccinium macrocarpon*	□ *Vaccinium oxycoccus*	□ *Viola epipsila*
2.2 Mire-specific mosses		
Instructions: Please mark all of the mire-specific mosses occurring within the peatland area.		
□ *Bryum longisetum*	□ *Calliergon stramineum*	□ *Calliergon trifarium*
□ *Calypogeia sphagnicola*	□ *Cephalozia connivens*	□ *Cephalozia macrostachya*
□ *Cephalozia pleniceps*	□ *Cephaloziella elachista*	□ *Cephaloziella spinigera*
□ *Cladopodiella fluitans*	□ *Dicranum bergeri*	□ *Drepanocladus cossonii,*
□ *Drepanocladus lycopodioides*	□ *Drepanocladus revolvens*	□ *Fissidens osmundoides*
□ *Hamatocaulis vernicosus*	□ *Helodium blandowii*	□ *Leiocolea rutheana*
□ *Lophozia laxa*	□ *Meesia hexastich*	□ *Meesia longiseta*
□ *Meesia triquetra*	□ *Meesia uliginosa*	□ *Mylia anomala*
□ *Paludella squarrosa*	□ *Pohlia sphagnicola*	□ *Polytrichum commune*
□ *Polytrichum strictum*	□ *Scapania paludicola*	□ *Sphagnum affine*
□ *Sphagnum angustifolium*	□ *Sphagnum balticum*	□ *Sphagnum capillifolium*
□ *Sphagnum centrale*	□ *Sphagnum compactum*	□ *Sphagnum contortum*
□ *Sphagnum cuspidatum*	□ *Sphagnum denticulatum var. denticulatum*	□ *Sphagnum denticulatum var. inundatum*
□ *Sphagnum fallax*	□ *Sphagnum fimbriatum*	□ *Sphagnum flexuosum*
□ *Sphagnum fuscum*	□ *Sphagnum magellanicum*	□ *Sphagnum majus*
□ *Sphagnum molle*	□ *Sphagnum obtusum*	□ *Sphagnum papillosum*
□ *Sphagnum platyphyllum*	□ *Sphagnum riparium*	□ *Sphagnum rubellum*

□ *Sphagnum spec.*	□ *Sphagnum subsecundum*	□ *Sphagnum tenellum*
□ *Sphagnum teres*	□ *Sphagnum warnstorfii*	□ *Splachnum ampullaceum*
□ *Tomentypnum nitens*	□ *Tomentypnum nitens*	
Total number of mire-specific vascular plants:		
Total number of mire-specific mosses:		

2.3 Classification for 'species diversity'

Number of mire-specific vascular plants:					Mosses present?		**Total score for species diversity**
≥ 10	≥ 7	≥ 4	≥ 1	0	Yes	No	Sum of points for mire-specific vascular plants and mosses
□ 4 points	□ 3 points	□ 2 points	□ 1 point	□ 0 points	□ 1 point	□ 0 points	**__ of 5 points**

3. Biocoenosis diversity

3.1 Habitat diversity

3.1.1 Mire-specific & typical plant formations *(Ellenberg & Müller-Dombois 1965, descriptions shortened).*
Remark: Species inventory and micro-relief as described in Ellenberg & Müller-Dombois 1966, but hydromorphologically no bogs are present in Brandenburg.

Instructions: Please mark all of the mire-specific and mire-typical plant formations that are present within the peatland area.

□	**Cold-deciduous swamp or peat forest (mainly broadleaved)** *(Flooded until late spring or early summer, relatively poor in tree species; ground cover mostly continuous; mainly broadleaved.)*
□	**Deciduous peat shrubland (or thicket)** *(Upright caespitose nano-phanerophytes with Sphagnum and (or) other peat mosses.)*
□	**Blanket bog** *(The microsurface of the bog is less undulating and less rich in actively growing mosses than in a typical raised bog. Scattered evergreen dwarf shrubs, caespitose hemicryptophytes (sedges or grasses) and some rhizomatous geophytes.)*
□	**Subcontinental woodland bog** *(Temporarily covered by low-productivity open woodland which, in a sequence of wetter years, may be replaced by Sphagnum formations similar to those of a typical raised bog.)*
□	**Tall sedge swamp (with creeping sedges)** *(Frequently flooded, often for long periods; as a rule natural. Foliage taller than 30–40 cm, sedges dominant throughout; creeping sedges forming large homogeneous stands, with very few other life forms.)*
□	**Tall sedge swamp (with caespitose sedges)** *(Frequently flooded, often for long periods; as a rule natural. Foliage taller than 30–40 cm, sedges dominant throughout; caespitose sedges forming tufts or hummocks, with very few other life forms.)*
□	**Low sedge swamp** *(Flooded only little or only for short periods, mostly anthropogeneous. Dominated by small sedges (Carex, Juncus, Scirpus, etc. with foliage no taller than 30 cm) of low productivity, intermixed with many other herbaceous life forms.)*
□	**Forb flushes (subdivision: calcareous)** *(Mostly dominated by small forbs - calcareous; older parts of plants covered by a white or brownish crust of precipitated carbonate.)*
□	**Forb flushes (subdivision: non calcareous)** *(Mostly dominated by small forbs - non-calcareous.)*
□	**Moss flush (subdivision calcareous)** *(Dominated by mosses – calcareous; older parts of plants covered by a white or brownish crust of precipitated carbonate)*
□	**Moss flush (subdivision non calcareous)** *(Dominated by mosses - non calcareous.)*
□	**Temperate and subpolar herbaceous floating meadows** *(Densely interwoven forbs covering permanent freshwater accumulations. Most of the phanerogams being halophytes, not true water plants. Herbacous floating meadow with pronounced seasonal aspects.)*
□	**Mossy floating meadow** *(Mainly mosses covering permanent freshwater accumulations. Mosses dominating throughout, but phanerogams may be present.)*

□	**Temperate and subpolar freshwater reedswamps** *(Mostly broadleaved plants which cannot endure high salt concentration. All shoots upright, only exceptionally floating in the water. In temperate and subpolar freshwater reedswamp, most plants yellow or dormant in winter.)*
□	**Temperate reedswamps on riverbanks** *(Shoots more flexible than in freshwater reedswamps or reedswamp formations of saltwater lakes. Sometimes with floating leaves.)*
Total number of mire-specific and mire-typical plant formations:	

3.1.2 Mire-specific and mire-typical special habitats

Instructions: Please mark all mire-specific and mire-typical special habitats present within the peatland area.

□ Lagg	□ Hummock	□ Hollow
□ Solitary trees	□ Running spring water	□ Upright dead wood
□ Lying dead wood	□ Open water body (temporary or permanent)	□ areas with no vegetation (e.g. mudbanks)
□ Mineral isles		
Total number of mire-specific and mire-typical special habitats:		

3.2 Habitat connectivity

3.2.1 Integration into biotope network

Instructions: The peatland biotope network of Brandenburg can be accessed at http://www.oeko-log.com/. For small mires and peatlands in forest, please choose the shapefile 'Small mires and peatlands in forests'; and for wet pastures and peatlands of the glacial valley, choose the shapefile 'Wet pastures and fens of the glacial valley'. The peatland needs to be part of a core or connecting area to be scored.

Chosen biotope network:	□ Small mires and peatlands in forests □ Wet pastures and fens of the glacial valley
Peatland is part of:	□ Core area □ Connecting area □ Development area □ Not part of biotope network

3.3 Classification for 'biocoenosis diversity'

Number of plant formations				Number of special habitats		Part of core or connecting area in biotope network?		**Total value**
≥ 5	≥ 3	≥ 1	0	≥ 3	≤ 2	Yes	No	Sum of plant formations, special habitats & integration into biotope network
□ 3 points	□ 2 points	□ 1 point	□ 0 points	□ 1 point	□ 0 points	□ 1 point	□ 0 points	**__ of 5 points**

4. Ecosystem diversity

In order to determine the dominant soil moisture class as well as the dominant degree of soil degradation, each site is subdivided into homogenous vegetation units (if there is more than one) firstly. Therefore areas with homogenous floristic dominances and physiognomic structure are segregated from each other. All units are outlined on recent satellite images and transferred into a geo information system to create spatial maps of each site. For each vegetation unit all plant species and their cover are recorded. To transfer this data into soil moistures classes, the water table indication of each plant species described by Koska 2001 is applied to determine the soil moisture class of each vegetation unit. Further, for each vegetation unit the upper 30 cm of topsoil peat are estimated in the field and classified into the different stages. Thereby also the dominant soil degradation can be spatially described.

4.1 Condition of topsoil peat

Instructions: For the condition of topsoil peat, please choose the spatially dominant stage.

	Stage	**Description**
□	Non-degraded peat (currently forming)	Peat of low decomposition ('fibric' (Joosten *et al.* 2017b)) in nutrient poor, acidic, base-rich or calcareous mires (e.g. moss peats, herbaceous peats with radicels and rhizomes); in naturally eutrophic mire ecosystems with or without natural water level fluctuations, such as alder forests, also moderately decomposed ('hemic') peat can occur (e.g. herbaceous peats with radicels and rhizomes or wood peat) (Schulz *et al.* 2019).
□	Non-degraded peat (currently not forming) or gyttja	Dry peat of low to moderate decomposition ('hemic' (Joosten *et al.* 2017b)) (divisions as above) as well as gyttja, meaning a sedentarily accumulated material that consist of at least 5 % (dry mass) of organic matter (Schulz *et al.* 2019).

□	Slightly degraded peat (highly decomposed peat)	Highly decomposed peat ('sapric' (Joosten *et al.* 2017b)); 'Compact, mainly homogeneous, dark brown to black mass; unstructured (amorphous) or aggregated into larger pieces; muddy to mushy consistency when wet, comparable to a squeezed-dry sponge when dry; no or a small amount of recognisable plant remains; plant remains usually limited to more highly decomposed wood or fibre fragments' (Schulz *et al.* 2019).
□	Moderately degraded peat (earthified peat)	'Dark brown to black-brown mass with crumb grain structure, consisting of bonded soil particles of various sizes (but mainly >1 mm); similar to garden mould; smeary consistency when wet, crumbly but never powdery-dusty when dry; no or only a small amount of recognisable plant remains' (Schulz *et al.* 2019).
□	Highly degraded peat (murshified peat)	'Black-brown to deep black, loose mass with fine granular structure, consisting of small (mainly <1 mm) bonded soil particles; thick, silty mass when very wet, smeary-granular when moist, distinctly granular and powdery-dusty when dry (resembling loose coal slack); no recognisable plant remains' (Schulz *et al.* 2019).
□	No peat	All soil substrates that are not peat (defined in Germany as sedentarily accumulated material that consists of more than 30 % (dry mass) of incompletely decomposed plant remains and humic substances or gyttja (Schulz *et al.* 2019).

4.2 Water table

Instructions: For the water table, choose the spatially dominant soil moisture class.

	Soil moisture class	**Water table relative to surface (+ above, - below)**	
		Long-term median water table in the wet season	**Long-term median water table in the dry season**
□	6+ (lower eulitoral)	+150 to +10	+140 to +0 cm
□	5+ (wet)	+10 to -5 cm	+0 to -10 cm
□	4+ (very moist)	-5 to -15 cm	-10 to -20 cm
□	3+ (moist)	-15 to -35 cm	-20 to -45 cm
□	2+ (moderately moist)	-35 to -70 cm	-45 to -85 cm
□	2- (moderately dry)	no value, water supply deficit <60 L m^{-2}	no value, water supply deficit <60 L m^{-2}

4.3 Classification for 'ecosystem diversity'

Topsoil peat / **Soil moisture class**	Non-degraded peat (currently forming)	Non- degraded peat (currently not forming) or gyttja	Slightly decomposed peat (highly decomposed peat)	Moderately degraded peat (earthified peat)	Highly degraded peat (murshfied peat)	No peat or gyttja
5+	□ 5	□ 5	□ 4	□ 4	□ 3	□ 3
4+ / 6+	□ 4	□ 4	□ 4	□ 3	□ 3	□ 2
3+	□ 4	□ 3	□ 3	□ 3	□ 2	□ 2
2+	□ 3	□ 3	□ 2	□ 2	□ 2	□ 1
2-	□ 3	□ 2	□ 2	□ 1	□ 1	□ 1
2- or lower	□ 2	□ 2	□ 1	□ 1	□ 0	□ 0

5. Classification for 'total mire-specific biodiversity'

Total points for species diversity:	
Total points for biocenosis diversity:	
Total points for ecosystem diversity:	
Sum:	

Class	**Accumulated points**	**Verbal description**	**Colour code**
5	14, 15	Very high mire-specific biodiversity	
4	11, 12, 13	High mire-specific biodiversity	
3	8, 9, 10	Moderate mire-specific biodiversity	
2	5, 6, 7	Low mire-specific biodiversity	
1	2, 3, 4	Very low mire-specific biodiversity	
0	0, 1	No mire-specific biodiversity	

3. Study II: Carbon content and other soil properties of near-surface peats before and after peatland restoration

3.1. Authors and affiliations

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3.2. Abstract

Peatland restoration usually aims at restarting the peatlands' function to store carbon within peat. The soil properties of the near-surface peat can give a first understanding of this process. Therefore, we sampled pH value, total organic carbon content (TOC), total nitrogen content (TN), C/N ratio as well as dry bulk density (BD), and describe the structure of near-surface peats in six restored fens in North-East Germany before (2002–2004) and after (2019–2021) restoration. Before restoration, the study sites showed peat degradation to various extents in their near-surface peats. PH values remained relatively stable over time. Comparing the degraded peat horizons, TOC increased significantly in four study sites, ranging from 35.7% to 47.8% in 2002–2004 and from 42.5% to 54.0% in 2019–2021. TN varied from 1.5% to 3.5% in 2002–2004 and from 1.8% to 3.2% in 2019–2021, but changes were only significant in one site, showing a slight decrease. In three sites, the increase in C/N ratio was significant, indicating lower nutrient availability. BD ranged from 0.08 to 0.48 g/cm3 in 2002–2004 and from 0.10 to 0.16 g/cm3 in 2019–2021, decreasing significantly in four sites. The structure of the degraded peat horizons changed after restoration to a more homogenous, sludge mass with larger re-aggregates. In three sites,

new peat moss peat layers above the degraded soil horizon were present in 2019–2021, with a mean thickness of 6.8 to 36.1 cm. The structure was comparable to typical, slightly decomposed peat moss peat. Our findings suggest that within about 17 years after fen restoration, and thereby a water table rise close to surface, TOC of the near-surface peats increased to values that are typical for undisturbed peatlands. This indicates that restoration can lead to the re-establishment of peatlands as potential carbon sinks, with TOC within the near-surface peat as one key factor in this process. Further, we assume that the decrease in nutrient availability, decrease of BD, and new, undisturbed peat layers can favor the establishment of mire-specific biodiversity and support ecosystem services similar to near-natural mires.

3.3. Key words

Mire, rewetting, pH value, nitrogen content, C/N ratio, dry bulk density

3.4. Introduction

Peatlands are a major global carbon store, containing more carbon than any other terrestrial ecosystem in vegetation and peat (Joosten et al. 2016). They are characterized by the presence of a naturally accumulated layer of peat, which is sedentarily accumulated material whose dry mass is at least 30% dead organic material, meaning 15% of total organic carbon, according to Joosten & Clarke (2002) and the German definition of mires (Ad-hoc-AG Boden 2005). Due to the water saturated and therefore anaerobic milieu in the soil, the decomposition of the on-site dead biomass is very slow and organic matter in the form of plant remains is stored as peat (Zeitz 2014a). The anaerobic conditions usually lead to the emission of methane (Lai 2009). If peatlands are drained, peat formation stops. Oxygen enters the near-surface peat and secondary pedogenetic processes, such as humification and mineralization are initiated, resulting in the release of the previously stored carbon and nitrogen as greenhouse gases carbon dioxide (CO_2) and nitrous oxide (N_2O) (Zeitz 2016).

Nearly 50% of the remaining European peatland area is degraded, in particular due to drainage for agriculture and forestry (Joosten & Tanneberger 2017). In the federal state of Brandenburg in North-East Germany, where the study area is located, only about 2% of the remaining 163.000 ha peatland area is in a natural state (LfU 2016; Luthardt 2014a). This results in estimated total emissions of 6.3 million t CO_2-Eq. per year, accounting for 11% of the state's annual greenhouse gas emissions (Reichelt 2021). The once existing specific biodiversity of mires (peatlands with a vegetation that actively forms peat (Joosten et al. 2017)) is lost, resulting for example in the decline of species, as 62% of mire-specific vascular plants in Brandenburg are listed as highly endangered, at risk of extinction or extinct (cf. LUA 2006; Luthardt 2014c; Hammerich et al. 2022). Ecosystem functions, such as stabilization of the landscape water balance or functioning as flood retention areas, water storage basins, groundwater nourishment areas and regulators of microclimate, are changed completely for the benefit of provisioning services (Luthardt & Wichmann 2016). In order to reduce CO_2 emissions and restore the function as carbon sinks, as well as biodiversity and other ecosystem functions of mires, peatland restoration efforts have risen (Bonn et al. 2016). Peatland restoration activities in Europe started in the late 20th century, but are estimated to be insufficient and slowly progressing (Roe et al. 2019; Tanneberger et al. 2021; Greifswald Mire Centre & Wetlands International 2022).

The near-surface peat is the location where carbon is accumulated for transfer into the long-term carbon store. Researching changes in carbon content in near-surface peat is therefore one key factor within this process (cf. Tolonen & Turunen 1996; Price et al. 2016).

With regard to peatlands as global carbon store, total organic carbon content (TOC) appears to be the most relevant soil parameter. TOC accounts for about 58% of the total organic matter formed by plant remains (Montanarella et al. 2006). Klingenfuß et al. (2014) specify this relation for peat soils in North-East Germany, giving values from 49% to 58%, depending on peat type.

The carbon to nitrogen (C/N) ratio is a key parameter regarding the trophic conditions in peatlands. The smaller the ratio, the more nutrients are plant-available (Succow & Stegmann 2001). In addition to organic matter, peat soils consist of varying proportions of mineral matter, also referred to as 'ash' (Succow 1988). This can involve allochthonous mineral matter that is carried in by surface waters, contributed by some form of soil amendment associated with agriculture or is attributable to the loss of organic matter during drainage-based peat mineralization (Klingenfuß et al. 2014).

Dry bulk density (BD) is an essential physical parameter and necessary to estimate peatland carbon stocks. Higher BD values are associated with a lower carbon content and increase from near-natural sites to sites impacted by drainage (Chapman et al. 2017; Wittnebel et al. 2021).

Due to the relatively short time period of peatland restoration activities and the restricted availability of long-term monitoring data (cf. Andersen et al. 2016), there is only little research focusing on carbon gained over time in the near-surface peats, especially regarding fens as geogenous peatlands in contrast to rainwater-fed bogs (Kotowski et al. 2016; Mrotzek et al. 2020).

In order to fill the knowledge gap concerning the re-establishment of restored peatlands, especially fens, as potential carbon sinks, we report changes in the soil parameters pH value, TOC, total nitrogen content (TN), C/N ratio, BD and structure of near-surface peats of six restored fens in North-East Germany before (2002–2004) and after (2019–2021) restoration measures were initiated. We refer to different hydrological and ecological site conditions as well as different soil degradation stages.

3.5. Materials and Methods

3.5.1. Study sites

The study sites, six fens in the nature park 'Stechlin-Ruppiner Land' (Table 3.1), are located in the federal state of Brandenburg, North-East Germany in the nemoral zone. The landscape formed in the late Pleistocene and is therefore characterized by large differences in height of the various landscape elements (high relief energy) and diversity of peatland types (Luthardt et al. 2002).

Table 3.1: Study sites, characteristics and restoration measures implemented in the EU-Life-Project 'Restoration of clear water lakes, mires and swamp forests of Lake Stechlin'. The data is modified from Luthardt et al. (2021). The classification of ecological and hydrogenetic peatland type follows Succow (1988), classification of peat degradation according to Schulz et al. (2019)

te and size (ha)	Ecological and hydrogenetic peatland type	Degra-dation of near-surface peat	Main restoration measures	Area share (%) of water table typical for near-natural mires		Trend of water table after resto-ration	Dominant vegetation after restora-tion on soil plots
				Before restora-tion (2002–2004)	After restora-tion (2019–2021)		
Beeren-wiese (10.5)	Mesotrophic to eutrophic subneutral terrestrialisation peatland	Earthi-fication	Water table rise in adjacent lake (+ 30 cm) (achieved in 2007)	49	51	Rising	Sedges (*Carex spec.*)
Boberow-seewiese (6.3)	Eutrophic subneutral percolation peatland	Murshi-fication	Partial closure of drainage ditch, water table rise in adjacent lake (+ 20 cm) (2003)	5	68	Rising	Sedges (*Carex spec.*)
Jäger-wiese (1.4)	Mesotrophic to eutrophic acidic terrestrialisation peatland	Beginning earthi-fication	Closure of drainage ditch (2003)	3	51	Rising	Sedges (*Carex spec.*) with few peat mosses (*Sphagnum spec.*)
Müller-wiese (2.0)	Mesotrophic to eutrophic acidic terrestrialisation peatland	Earthi-fication	Closure of drainage ditch (2003)	15	15	Rising	Peat mosses (*Sphagnum spec.*)
Seggen-kuten (0.17)	Mesotrophic to eutrophic acidic water rise peatland	Beginning earthi-fication	Removal of marginal spruces, stocking reduction in the above ground catchment area (2003)	45	100	Rising	Peat mosses (*Sphagnum spec.*)
Teufels-seemoor (0.95)	Mesotrophic to eutrophic acidic kettle hole peatland	Beginning earthi-fication	Closure of drainage ditch, pine curling, removal of marginal spruces (2003)	24	100	Rising	peat mosses (*Sphagnum spec.*)

The ‘hydrogenetic peatland type’ provides information on the hydrological conditions of formation and the resulting composition of the peatland. Thereby, Succow (1988) distinguishes between ombrogenous bogs and seven different geogenous fen types, a classification largely congruent to the suggestions given by Joosten & Clarke (2002) for a global hydrogenetic classification of peatlands. The study sites comprise three terrestrialization peatlands (Beerenwiese, Jägerwiese, Müllerwiese), one percolation peatland (Boberowseewiese), one water rise peatland (Seggenkuten) and one kettle hole peatland (Teufelsseemoor) (Table 3.1). The specifications regarding the ‘ecological peatland type’ given in Table 3.1 also follow Succow (1988). These types provide information on the chemical quality of the feeding water in terms of trophic conditions (nutrient availability) and base saturation (acidity), which lead to the establishment of different characteristic plant communities under undisturbed conditions. Regarding the trophic conditions, which are quantified by the C/N ratio of the peat, Succow (1988) distinguishes oligotrophic (>33), mesotrophic (33–20) and eutrophic (<20–10) peatlands. The base saturation is deviated from the pH value of the peat: acidic (<4.8), subneutral (4.8–6.4) and calcareous (>6.4). The study sites comprise a variety of ecological types: four of them are meso-to eutrophic acidic (Jägerwiese, Müllerwiese, Seggenkuten, Teufelsseemoor), one is meso- to eutrophic subneutral (Beerenwiese), and one is eutrophic subneutral (Boberowseewiese).

The long-term annual rainfall in the study area is 528 mm, while the long-term annual mean temperature is 9.3 °C (period 2002-2021). In comparison to the 30-year average from 1961–1990, the temperature increased by 1.2 K and the precipitation decreased by 59 mm (DWD 2022). The climatic water balance is negative, so that the water content of the peatlands is always dependent on inflowing water from the surrounding environment (Luthardt 2014).

Until 2002, all six fens were drained, most of them directly by drainage ditches, some also by indirect drainage due to evaporation-intensive stocking of conifers in the aboveground catchment area or lowering of the water table of adjacent lakes.

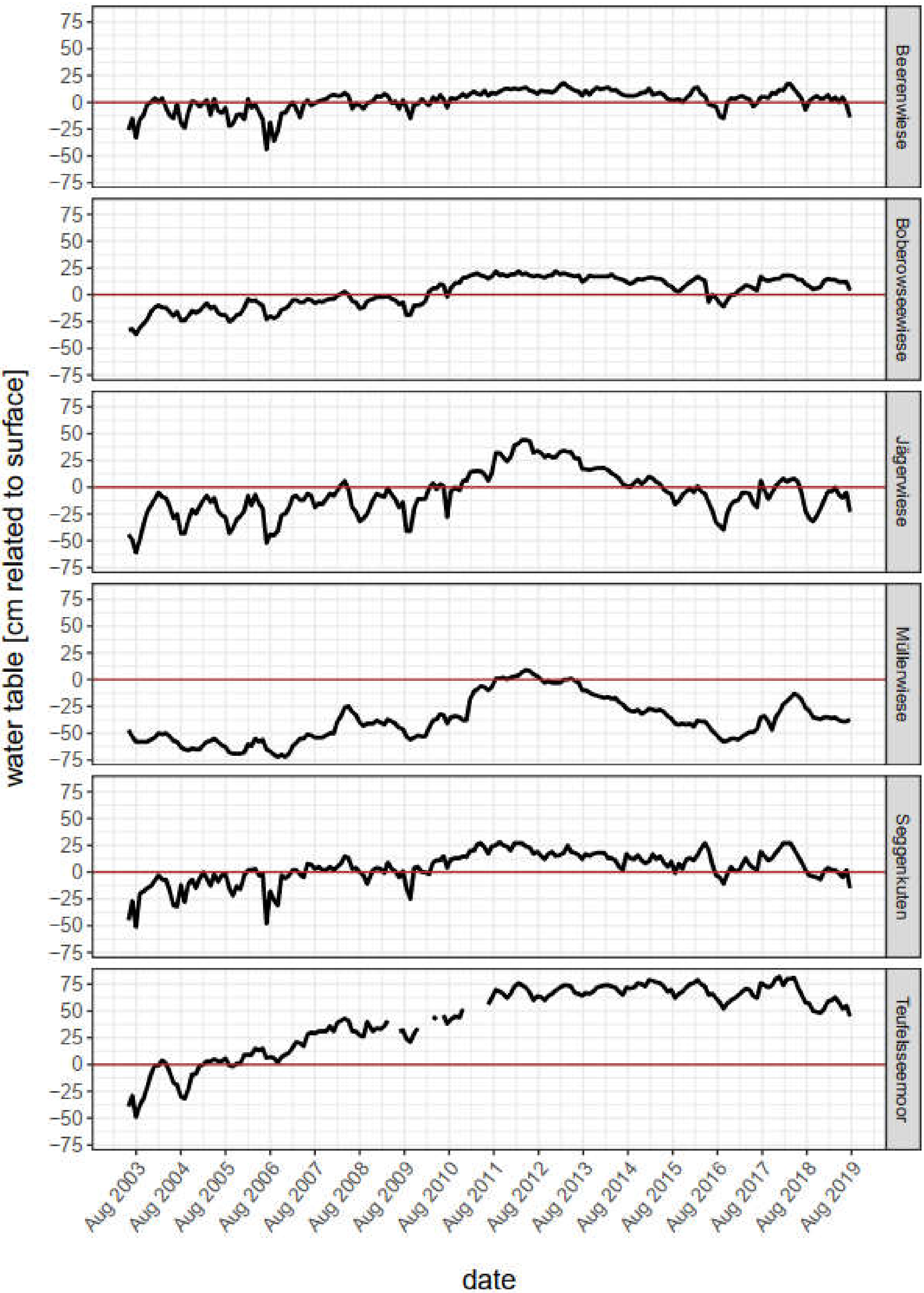

Figure 3.1: Water tables in the study sites from 2002 (before restoration) to 2019 (cm related to peatland surface = 0).

Water tables in all fens were several tens of centimeters below peatland surface. Consequently, near-surface peats in all sites exhibited different stages of oxygen-induced soil degradation. The natural peat structure changes gradually to a crumb or even fine granular structure with substantially changed soil properties (Ilnicki & Zeitz 2003). In German soil classification, moderate drainage leads to the development of 'earthified' peat, characterized by a crumb grain structure resembling garden mold. Under intensive drainage, aeration and ongoing degradation, the crumb structure subsequently changes into a structure of fine granular soil particles, referred to as 'murshified' peat (Ad-hoc-AG Boden 2005; Schulz et al. 2019).Between 2002 and 2004, several restoration measures were initiated as part of the EU Life-Project 'Restoration of clear water lakes, mires and swamp forests of the Lake Stechlin' (Project-Ident: LIFE00 NAT/D/007057) (Luthardt et al. 2002; Luthardt et al. 2021), for details of restoration measures see Table 3.1. The water table was continuously measured monthly since June 2003 in the center of each fen in a permanently installed water level tube with a water level meter. All sites exhibit a water table rise following restoration (Figure 3.1), due to the increased water retention as a consequence of the different restoration measures. A maximum was reached between 2011 and 2012, after years with high precipitation (2007, 2010, 2011). The percentage of area with a water table typical for near-natural mires (annual mean at least at peatland surface or higher) increased distinctly in four of the six sites about 15 years after restoration (Table 3.1). The highest increase was recorded in Teufelsseemoor with a peak water table up to 75 cm above surface, leading to a temporary shallow water body. In contrast, Müllerwiese is the only site where the water table is mainly still below peatland surface and the closure of the drainage ditch could not yet compensate for decades of drainage. Further, potentially peat-forming plant species which are adapted to high water tables, mainly sedges (*Carex* spec.) in Beerenwiese, Boberowseewiese, Jägerwiese or mainly peat mosses (*Sphagnum* spec.) in Müllerwiese, Seggenkuten, Teufelseemoor, could re-establish or expand (Figure 3.2) (Luthardt et al. 2021)

3.5.2. Soil sampling

The field research was approved by the administration of nature park 'Stechlin-Ruppiner Land' as representative of the State Environmental Agency of Brandenburg. Soil data collection before restoration was conducted in 2002 and 2004 (Beerenwiese, Boberowseewiese, southern part of Seggenkuten, Teufelsseemoor in 2002; Jägerwiese, Müllerwiese, northern part of Seggenkuten in 2004). On each site, we installed two soil measurement fields with five sampling plots, respectively, giving ten replicas per site. Sampling plot locations were recorded via precision-GPS (LEICA GS 50). At each plot, we collected soil samples in the uppermost homogenous soil horizon, the degraded near-surface peat in a depth of 0–20 cm. Samples were taken volumetrically with sample rings of 100 cm^3 volume and additionally a non-volumetrical bag sample of about 500 g at the same depth for further analysis.

Repeated, analogous sampling was conducted in 2019 for Beerenwiese, Boberowseewiese, Jägerwiese, and in 2021 for Müllerwiese, Seggenkuten and Teufelsseemoor. In some of the peat moss-dominated plots, we found a newly grown peat moss peat layer above the degraded peat, which formed the original surface layer in 2002–2004. In all cases, the newly formed peat moss peat was clearly distinguishable from the degraded peat layer below by color and structure (compare Figure 3.3), appearing as a sharp border. While sedge roots as peat forming plant parts grow vertically into existing peat and form 'displacement peat', peat mosses grow upwards and form a new layer of peat moss peat above the existing peat when dying (Michaelis et al. 2020). In plots with a newly grown peat moss peat layer, we additionally collected soil samples from this new layer. The comparative samples to 2002–2004 were collected in the layer below, which corresponds to the original surface layer in 2002–2004. For Teufelsseemoor, volumetrically sampling was not possible due to the distinct fluffiness of the newly formed peat, which did not allow cutting without distorting its shape.

Figure 3.2: Photo comparison of study sites before and after restoration.

Figure 3.3: Representative peat monoliths of displacement peat and newly formed peat moss peat. A: wet and re-aggregated displacement peat formed by ingrowing and dying sedge roots in the degraded near-surface peat after restoration. B: wet and re-aggregated degraded near-surface peat after restoration with 10 cm of newly formed peat moss peat on top, distinguishable by colour and structure.

3.5.3. Laboratory analysis of soil parameters

Analysis of soil parameters were conducted at the analytical laboratory of Eberswalde University for Sustainable Development, Germany.

The determination of pH value was conducted according to DIN EN ISO 10390 (2005): composition of air-dry samples (dried at 40 °C in a forced-air oven) with calcium chloride solution and measurement after 1 h with an electrode (WTWMulti1970i).

Determination of BD was carried out according to DIN EN ISO 11272 (2017): samples were dried at 105 °C in a forced-air oven to constant mass. BD (g/cm^3) was calculated by dividing the dry mass (g) by the volume of the sample ring (100 cm^3) (Chambers et al. 2011).

Determination of total carbon content (TC) content was conducted according to DIN EN 15936 (2022) and TN according to DIN 13878 (1995). Samples were dried at 40 °C and 1 to

2 g substrate were grounded in a fine mill to powder and dried again at 105 °C as pretreatment for dry combustion (1,200 °C) with an elemental analyzer (LECO Trumac CNS). Due to low pH values and conduction of a carbonate pre-test with 10% hydrochloric acid, all samples were to be regarded as carbonate-free. Therefore, the measured TC corresponds to TOC.

3.5.4. Statistical analysis of soil parameters

For each site and each time, mean values and standard deviations were calculated for the soil parameters pH value, TOC, TN, C/N ratio and BD. The Wilcoxon-test was used to detect possible significant differences of soil properties regarding time span before and after restoration, since not all samples were normally distributed, indicated by the ShapiroWilk-test. All statistics were carried out with R (R Core Team 2023), package 'psych' (Revelle 2023) and 'EnvStats' (Millard 2013). Figures were drawn with R-package 'ggplot2' (Wickham 2016).

3.6. Results

All mean values of soil parameters described in the following refer to the comparison of the degraded peat layer before and after restoration about 17 years later. The parameters are given in detail in Table 3.2 and Figures 3.4 to 3.8.

In five of the six sites, pH values remain relatively stable over time, ranging from 3.1 to 3.6 in the acidic fens and from 5.0 to 5.4 in the subneutral ones. Only in Beerenwiese, there is a slight but significant decrease from 5.4 to 5.2 after restoration (Figure 3.4).

TOC between sites ranges from 35.7% to 47.8% in 2002–2004 before restoration and from 42.5% to 54.0% in 2019–2021 after restoration. TOC increases distinctly in five sites (four significantly, thereof three sedge-dominated, one peat moss-dominated) and remains stable in Müllerwiese. The highest gain (15.1%) is recorded in Teufelsseemoor (Figure 3.5).

Differences in TN are less distinctive and only significant for one site (Beerenwiese decrease from 3.5% to 3.2%). The TN decreases in four sites, increases in two sites and ranges from 1.5% to 3.5% in 2002–2004 and from 1.8% to 3.2% in 2019–2021 after restoration (Figure 3.6).

The C/N ratio increases in four sites (three significantly), indicating a lowered nutrient availability. These four sites are the same with a recorded decrease in TN (Beerenwiese, Boberowseewiese, Seggenkuten, Teufelsseemoor) (Figure 3.7).

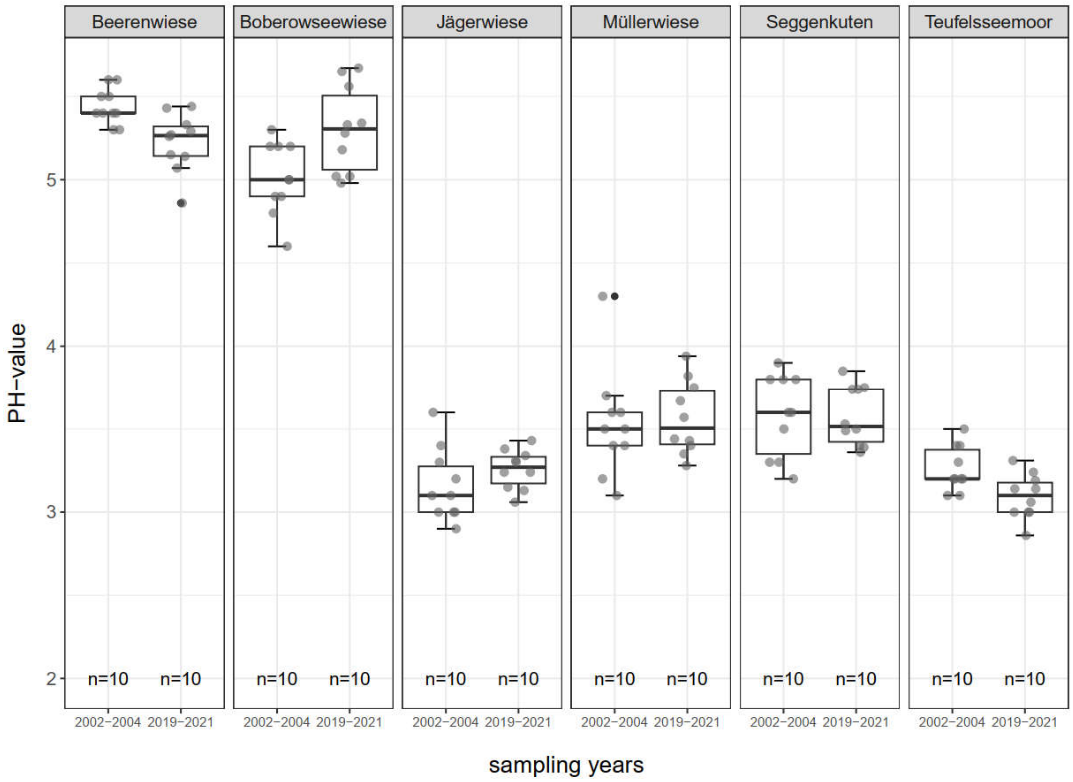

Figure 3.4: PH value of the degraded near-surface peats before (2002–2004) and after (2019–2021) restoration.

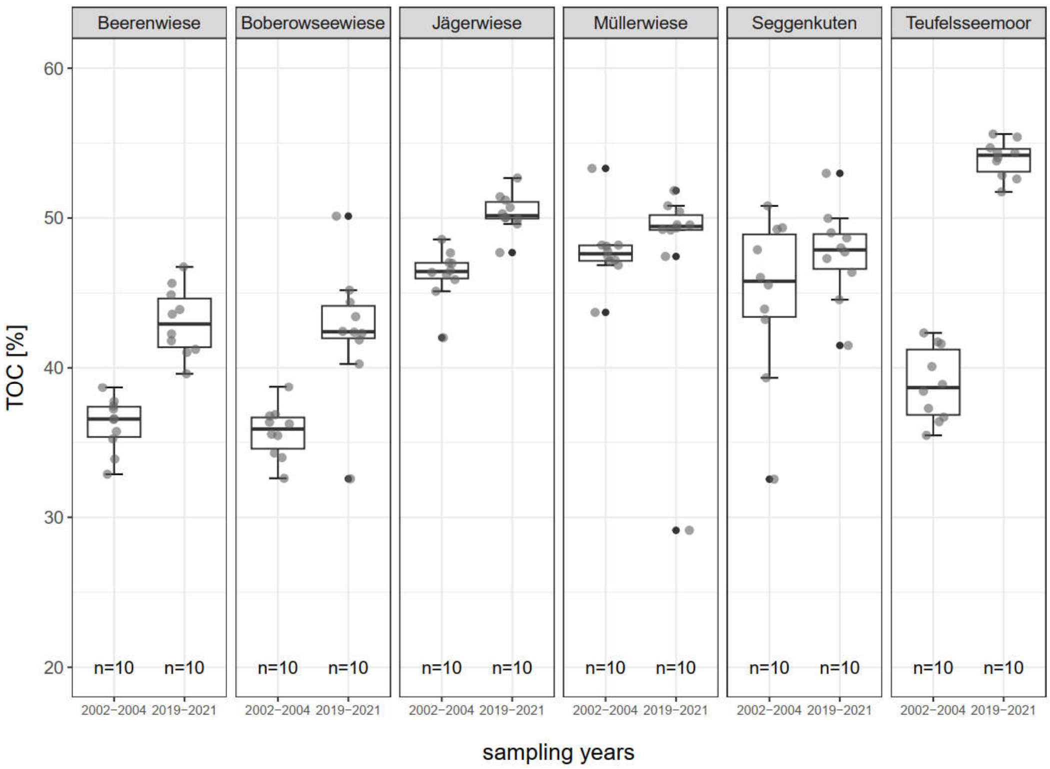

Figure 3.5: TOC-total organic carbon (%) of the degraded near-surface peats before (2002–2004) and after (2019–2021) restoration.

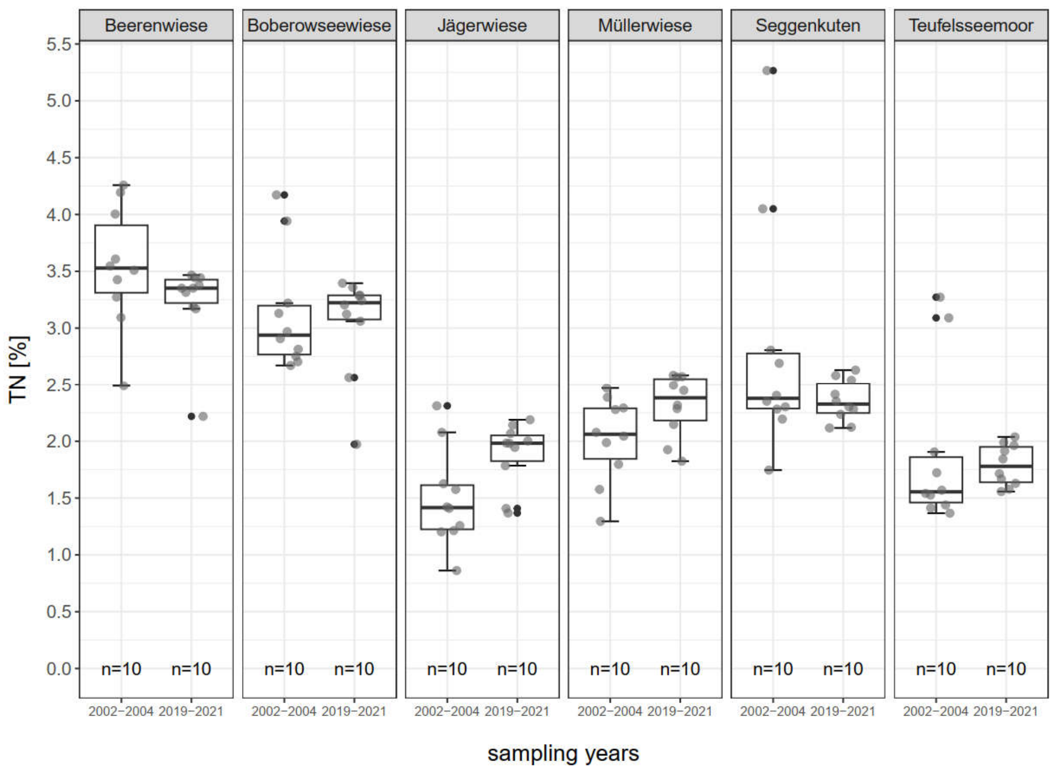

Figure 3.6: TN-total nitrogen (%) of the degraded near-surface peats before (2002–2004) and after (2019–2021) restoration.

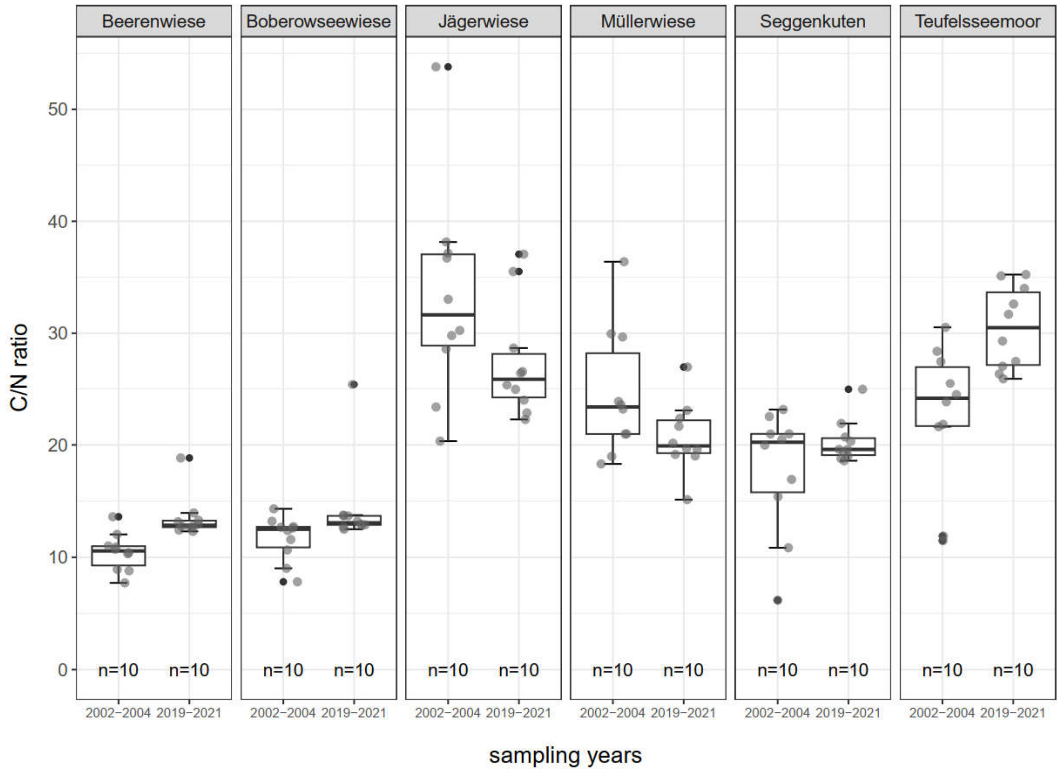

Figure 3.7: C/N ratio of the degraded near-surface peats before (2002–2004) and after (2019–2021) restoration.

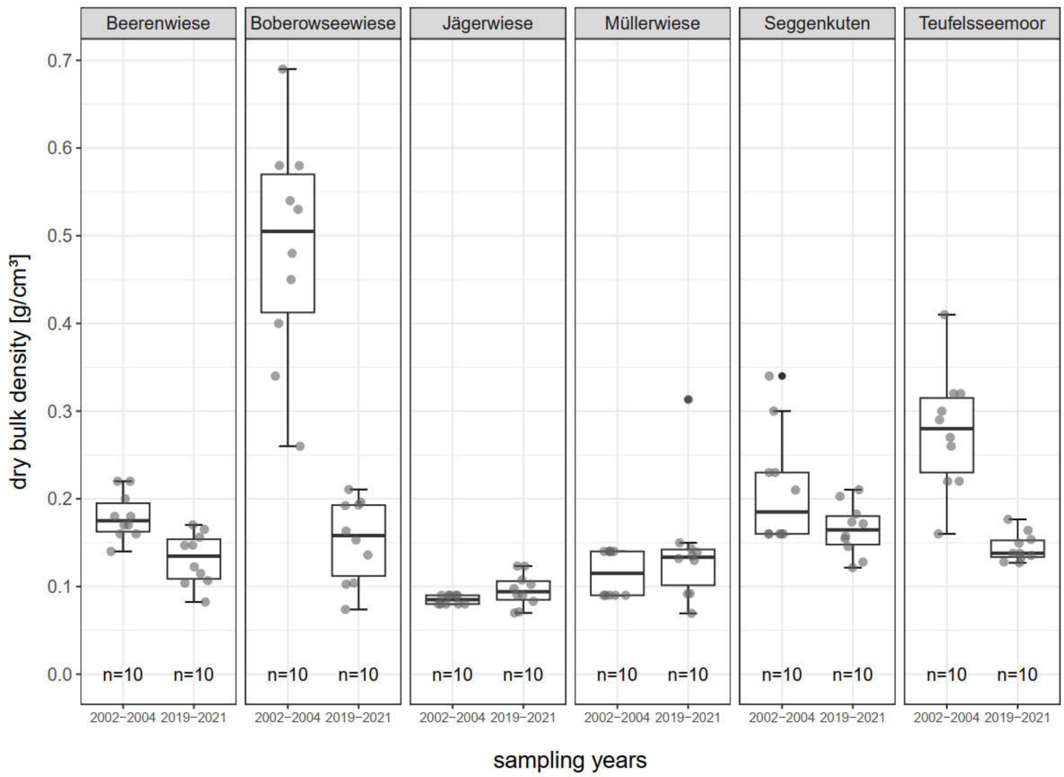

Figure 3.8: BD-dry bulk density (g/cm^3) of the degraded near-surface peats before (2002–2004) and after (2019–2021) restoration.

Table 3.2: PH value, TOC, TN, C/N ratio and BD of the degraded near-surface peats before (2002–2004) and after (2019–2021) restoration. Mean values, standard deviations (SD, in brackets) and respective number of replicas for the soil parameters pH value, TOC, TN, C/N ratio and BD of the degraded near-surface peats of the study sites before (2002–2004) and after (2019–2021) restoration. Asterisks indicate significant changes ($p < 0.05$) between both time series.

Site	Sampling year	Number of replicas	PH value	TOC - total organic carbon (%), mean (SD)	TN - total nitrogen (%), mean (SD)	C/N ratio, mean (SD)	BD - dry bulk density (g/cm³), mean (SD)
Beeren-wiese	2002–2004	10	5.44 (0.11)	36,18 (1,79)	3.54 (0.53)	10,44 (1,69)	0.18 (0.03)
	2019–2021	10	5.22 (0.18) *	43,06 (2,26) *	3.23 (0.37) *	13,50 (1,93) *	0.13 (0.03) *
Boberow-seewiese	2002–2004	10	5.01 (0.22)	35,67 (1,72)	3.13 (0.52)	11,69 (1,99)	0.48 (0.13)
	2019–2021	10	5.30 (0.26)	42,49 (4,40) *	3.05 (0.45)	14,36 (3,91) *	0.15 (0.05) *
Jäger-wiese	2002–2004	10	3.16 (0.22)	46,23 (1,76)	1.50 (0.43)	33,12 (9,31)	0.08 (0.01)
	2019–2021	10	3.26 (0.12)	50,36 (1,31) *	1.89 (0.29)	27,36 (5,07)	0.10 (0.02)
Müller-wiese	2002–2004	10	3.53 (0.33)	47,79 (2,34)	2.02 (0.37)	24,59 (5,72)	0.12 (0.03)
	2019–2021	10	3.56 (0.22)	47,65 (6,61)	2.32 (0.27)	20,68 (3,12)	0.14 (0.07)
Seggen-kuten	2002–2004	10	3.58 (0.25)	44,79 (5,49)	2.81 (1.05)	17,75 (5,51)	0.21 (0.07)
	2019–2021	10	3.58 (0.18)	47,61 (3,09)	2.36 (0.18)	20,28 (1,92)	0.16 (0.03) *
Teufelssee-moor	2002–2004	10	3.26 (0.13)	38,88 (2,46)	1.88 (0.70)	22,70 (6,44)	0.28 (0.07)
	2019–2021	10	3.09 (0.13)	53,95 (1,23) *	1.79 (0.18)	30,48 (3,69) *	0.14 (0.02) *

BD between sites ranges from 0.08 to 0.48 g/cm³ in 2002–2004 and from 0.10 to 0.16 g/ cm³ in 2019–2021. It significantly decreases in four sites (Beerenwiese, Boberowseewiese, Seggenkuten and Teufelsseemoor), most remarkably in Boberowseewiese with the formerly highest degree of peat degradation (murshification) from 0.48 g/ cm³ before rewetting to 0.15 g/ cm³ afterwards. There are slight and not significant increases in Jägerwiese (0.08 to 0.10 g/cm3) and in Müllerwiese (0.12 to 0.14 g/cm³) (Figure 3.8).

The typical structure of the formerly degraded peat horizons (crumb grain structure for earthified peat and fine granular structure for murshified peat according to Schulz et al. 2019) changed to a more homogenous, sludged mass with larger re-aggregates in the rewetted peats.

In three sites, the formation of a new peat layer formed by peat mosses above the degraded soil horizon was recorded in 2019–2021 (nine soil plots in Müllerwiese, four in Seggenkuten and all ten plots in Teufelseemoor (Table 3.3). The mean thickness of this layer ranges from 6.8 cm in Seggenkuten to 36.1 cm in Teufelsseemoor. This is equivalent to mean accumulation rates of 0.38–2.01 cm/y considering the time since restoration measures in 2003. The structure of the newly formed peat moss peat is analogous to slightly decomposed peat moss peat, as described by Schulz et al. (2019): loosely bedded, felty and well preserved moss plants with a characteristic straw yellow to light brown color (Figure 3.3). Mean TOC in new peat is comparable to TOC in older peat at two of the three sites after restoration, whereas in Teufelsseemoor it is significantly higher in the new peat (51.7%). TN is significantly lower in one site (Müllerwiese), significantly higher in one site (Teufelsseemoor) and remains stable in Seggenkuten. Values for C/N ration act reciprocal (Table 3.3). BD in this new peat is lower in all measured sites, but only significantly in Müllerwiese.

Table 3.3: Thickness, pH value, TOC, TN, C/N ratio and BD of the newly formed peat moss peat above the degraded peat of the study sites Müllerwiese, Seggenkuten and Teufelsseemoor in 2021. Mean values, standard deviations (SD, in brackets) and respective number of replicas for thickness and the soil parameters pH value, TOC, TN, C/N ratio and BD of the newly formed peat moss peat above the degraded peat of the study sites Müllerwiese, Seggenkuten and Teufelsseemoor in 2021. Asterisks indicate significant changes ($p < 0.05$) between newly formed peat and degraded peat beneath.

Site	Number of replicas	Thickness of newly formed peat moss peat above degraded peat (cm), mean (SD)	PH value	TOC - total organic carbon (%), mean (SD)	TN - total nitrogen (%), mean (SD)	C/N ratio, mean (SD)	BD - dry bulk density (g/cm^3), mean (SD)
Müller-wiese	9	15.78 (5.89)	3.24 (0.10)*	48.83 (0.51)	1.33 (0.27)*	37.96 (7.21)*	0.06 (0.02)*
Seggen-kuten	4	6.75 (4.27)	3.45 (0.19)	48.14 (1.02)	2.25 (0.50)	22.57 (6.68)	0.13 (0.07)
Teufels-seemoor	10	36.10 (8.58)	3.36 (0.17)*	51.69 (1.93)*	1.96 (0.14)*	26.46 (2.48)*	n.s.

3.7. Discussion

There are only a few comprehensive studies in North-East Germany regarding values for TOC in degraded peat soils. Succow (1988) determines a mean of 35.0% (span: 7.2 to 49.6, standard deviation: 12.5) for earthified peats. Kühn et al. (2015) find a comparable mean of 31.4% for earthified peats and a lower mean of 27.5% for murshified peats, since organic carbon decreases with increasing soil aeration and degradation. With the exception of Beerenwiese, the mean values for TOC in the study sites before restoration are higher than the comparative values given in literature. This is probably due to the fact, that these fens have been drained only moderately during usage as extensive grassland and therefore, there was only slight topsoil degradation and loss of organic carbon in most of the study sites. Three of the six fens showed only a beginning earthification before rewetting (Table 3.1). The non-organic mass of the six fens is mainly attributable to the loss of organic content during peat degradation since there are no surface waters carrying in allochthonous mineral material. Furthermore, the above ground catchment areas have been forested for the past decades, so that soil erosion into the fens is negligible.

The most comprehensive compilation of Holocene peat soil properties for the northern hemisphere is given by Loisel et al. (2014), whereby ombrotrophic bogs are more strongly represented than minerotrophic fens (20% of all sites), only few plots are located in Germany, and the focus is on complete peat cores from surface to peatland base. Loisel et al. (2014) determine a mean TOC of 47.4% (n = 96) for 'humified' peat, as highly decomposed peat which is not assignable to a specific plant group. Succow (1988) found a mean of 40.8% TOC for non-degraded sedge peat in North-East Germany. The TOC values for sedge-dominated sites (Beerenwiese, Boberowseewiese, Jägerwiese) in the near-surface peats are higher (42.5% to 50.4%) about 17 years after restoration. Loisel et al. (2014) give a mean of 50.5% TOC in northern peatlands (n = 519) regarding 'herbaceous' peat (including peats formed by sedge roots), which is reached by one of the sites in 2019–2021. For peat moss peat, Succow (1988) found a mean of 48.6% TOC. Values for degraded near-surface peats in peat moss-dominated sites are comparable for two sites (Müllerwiese: 47.7%, Seggenkuten: 47.6%) and are higher in Teufelsseemoor: 54.0% in 2019–2021. All three sites show higher TOC than the mean value for 'Sphagnum' peats in the northern hemisphere: 46.0 (n = 1520) as given by Loisel et al. (2014) after restoration. But, the comparison of TOC of the near surface peat with TOC of deeper layers must be interpreted cautiously, as the deeper layers have been undergoing continuous decomposition and the newly added material and TOC will only partially or even never become part of the long-term carbon store (cf. Young et al. 2019).

The distinct increase of TOC in five study sites (four significantly, whereof three sedge-dominated, one peat moss-dominated) after rewetting as well as the comparison with carbon contents of non-degraded sedge and peat moss peat indicate that only a few decades after restoration, an organic carbon content typical for undisturbed peatlands can be regained in degraded peat horizons. Prerequisite for these developments is a water table typical for near-natural mires allowing for accumulation of peat and the re-establishment of potentially peat forming plant species. These potentially peat forming species, such as *Sphagum* spec. and most *Carex* species are not only dependent on wet conditions to be competitive, but a high water table is a requirement to allow dying plant parts of these species to form peat and not be oxidated. The divergent development of TOC in Teufelsseemoor and Müllerwiese emphasizes the impact of the hydrological conditions on carbon storage: Teufelsseemoor with the highest increase in water table shows the highest gain in TOC in near-surface peats after restoration whereas Müllerwiese with a water table mainly still below peatland surface and ongoing aeration of peat shows no gain in TOC.

Research on the restoration of fen systems indicates, that the lower the degree of initial degradation, the higher the potential for restoring fens to near natural conditions (cf. Grootjans & Van Diggelen 1995; Klimkowska et al. 2010). However, research so far focused mainly on the development of typical, sometimes highly endangered plant species, not on soil properties. Our results do not indicate that a lower initial degree of soil degradation leads to a higher restoration success concerning TOC, since we determined a significant gain in TOC in sites with a beginning earthification (Jägerwiese, Teufelsseemoor), an earthification (Beerenwiese) as well as a murshification (Boberowseewiese) before restoration. We recommend additional research with a higher number of study sites with different soil degradation stages to further investigate this debate.

Regarding sedge-dominated fens, the input of new carbon happens by ingrowing and later dying sedge roots (Michaelis et al. 2020). In the peat moss-dominated fens of our study, new peat of dying peat mosses is deposited on top of the degraded peat. A possible explanation for carbon enrichment in lower soil horizons may be the deposition of dissolved organic carbon released in case of high water tables. Mrotzek et al. (2020) and Michaelis et al. (2020) report a newly deposited material after 20 years of restoration of roots, radicals and litter, and describe it as a first accumulation of litter leading to a so-called 'proto-peat'. This indicates that also in sedge-dominated peatlands, new deposits on degraded peat soil are possible within a time-frame comparable to our study.

Still, TOC values of our study within the near-surface peats cannot be directly transferred to describe long-term carbon storage. Due to ongoing decay in the highly dynamic, partially not water saturated near-surface peat ('acrotelm'), our results can only be used to describe the potential carbon to be transferred into the permanently water saturated, long term carbon store ('catotelm'), where decomposition is very slow (Tolonen & Turunen 1996; Young et al. 2019). This process is influenced by various factors such as climate, water table, vegetation and peatland type (cf. Tolonen & Turunen 1996; Malmer et al. 1994; Charman et al. 2015; Milner et al. 2020). Young et al. (2019) show, that using changes in carbon accumulation in near-surface peats for estimating long-term carbon storage might be misleading and should not be used as the only source of evidence of management strategies on peatland carbon. Especially in the study area, due to the climatic change with higher temperatures and decreased precipitation, negative climatic water balance (cf. DWD 2022) and therefore high dependency of the peatland on inflowing water from the surrounding environment, the development of the carbon transferred into long-term-store is unclear. However, the water tables in our study sites are relatively stable after restoration also in years of low precipitation as in 2018. This could be a hint for increasing resilience of the peatland systems after restoration by increasing porosity and water storage capacity of the near-surface peats.

Regarding content of TN, there are also only few comprehensive studies in Germany. Grosse-Brauckmann (1990) and Naucke (1990) state values of 2.0% to 4.0% and respectively 2.5% to 4.5% for upper fen soils in general. Feige (1977) gives contents of 2.47% to 3.07% for undisturbed sedge peat and 0.5% to 1.33% for undisturbed peat moss peat. Loisel et al. (2014) give for northern peatlands a mean of 1.5% (n = 96) for 'humified' peat, a mean of 1.7% (n = 518) for 'herbaceous' peat and a mean of 0.7 (n = 1,523) for 'Sphagnum' peat. TN values of our study sites are in line with comparative values in

Germany but higher than values given for the northern hemisphere, presumably due to the high share of ombrotrophic bogs having lower TN contents than fens which are supplied with mineral water. The detected relative TN decrease in four sites (only one significant) after rewetting can be mainly attributed to the gain of carbon, as described by Malmer & Holm (1984), Ohlson & Økland (1998) or Turunen et al. (2001). Malmer & Wallén (2004) assume that nitrogen is conserved quantitatively in the peat because of hardly any losses during decay. This is also apparent regarding the C/N ratio. The ratio increases in four sites (three significantly), indicating a lowered nutrient availability. Again, this can be mainly attributed to the gain of carbon after rewetting. These findings are of high importance for specific fen plant species, with a high share of adaption to rather mesotrophic conditions.

Mean values for BD in North-East Germany are given by Kühn et al. (2015) for earthified peats: 0.3 g/cm^3 and murshified peats: 0.5 g/cm^3 as well as by Schindler et al. (2003) for earthified peats: 0.28 g/cm^3 and murshified peats: 0.42 g/cm^3. Values are comparable to our sites before restoration, although some sites show a lower BD due to an only beginning earthification. The significant decrease of BD after rewetting in four sites can be attributed to the gain in carbon, as there is a clear relation between low BD and high carbon content (Chapman et al. 2017; Wittnebel et al. 2021). The most remarkable decrease of BD in Boberowseewiese, in particular, can only be fully explained by the combination of the carbon increase due to ingrowing and dying sedge roots and the loosening of peat by buoyancy after resaturation. Values after restoration in the sedge-dominated sites are in line with values given by Loisel et al. (2014) for 'herbaceous' peat in northern peatlands with a mean of 0.12% (n = 3,188). The values for our peat moss-dominated sites are about double as the mean of 0.08 (n = 4,372) for 'Sphagnum' peat given by Loisel et al. (2014).

PH values remain relatively stable before and after restoration, since quantity but not quality of the feeding water has changed by rewetting.

3.8. Conclusion

Fen restoration with re-establishment of water tables typical for undisturbed peatlands, with regard to our six study sites in North-East Germany, leads to stable pH values, increased TOC comparable to contents of undisturbed peatlands, decreased nutrient availability in terms of C/N ratio, and decreased BD (resembling values of undisturbed peat for sedge-dominated sites) in the degraded, near-surface peats.

Further, as for mire-specific biodiversity described by Hammerich et al. (2022) not only on the level of species, but with peat accumulation and water table close to surface as determining eco-hydrological properties on the ecosystem level, the regeneration of the near-surface peat can also be seen as central for re-establishing the biodiversity of mires.

3.9. Acknowledgements

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3.10. Author contributions

Jenny Hammerich conceived and designed the experiments, performed the experiments, analyzed the data, prepared figures and/or tables, authored or reviewed drafts of the article, and approved the final draft; Corinna Schulz conceived and designed the experiments, performed the experiments, analyzed the data, prepared figures and/or tables, authored or reviewed drafts of the article, and approved the final draft; Robert Probst conceived and designed the experiments, performed the experiments, analyzed the data, prepared figures and/or tables, and approved the final draft; Thomas Lüdicke performed the experiments, prepared figures and/or tables, and approved the final draft; Vera Luthardt conceived and designed the experiments, authored or reviewed drafts of the article, and approved the final draft.

4. Study III: Monitoring peatland restoration in forests – The effects of hydraulic and management measures on the water table, peat accumulation, mire-specific biodiversity and greenhouse gas emissions

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4.2. Abstract

Peatland restoration aims to maintain and re-establish mire biodiversity and their important ecosystem functions, such as carbon storage and landscape water regulation. Restoration thereby affects diverse peatland properties differently due to varying site conditions and associated factors. Herein, we present monitoring results for 33 restored peatlands in north-east Germany. Data were collected prior to – and four to 27 years after – the initiation of restoration measures. By using vegetation as a proxy for on-site conditions, alongside the additional sampling of near-surface peats and the application of established assessment tools, we determined water tables and peat-accumulating area, assessed mire-specific biodiversity and approximated greenhouse gas (GHG) emissions. Peatlands drained by ditches were generally in a higher initial degraded state. In this regard, hydraulic measures led to a significant rise in the water table, the re-initiation of peat accumulation, enhanced mire-specific biodiversity and savings in GHG emissions, albeit not always comparable to near-natural conditions. Peatlands not drained by

ditches but showing early signs of degradation have undergone management measures such as the removal of tall vegetation and sapling trees, and forest restructuring in the peatland catchment area. In contrast, these peatlands were in a better initial state, and management measures resulted in preserving pre-measurement conditions. Groundwater-abstracting facilities in the peatland catchment area significantly hindered restoration success. Principal component analysis showed positive relations between restoration success and years since restoration, whereas the acid-base ratio and trophic condition of the peatland before restoration, the hydrogenetic mire type and peatland size did not correlate with restoration success.

4.3. Implications for practice

- In terms of climate change and biodiversity loss, the top priorities of nature conservation need to be protecting and stabilising mires and restoring peatlands.
- Forest restructuring to near-natural conditions within peatland catchment areas is essential for conserving and restoring peatlands in forests. It should also be considered for remaining mires, as the effects are long-lasting and must be implemented immediately to counteract future reduced water availability due to climate change.
- We recommend the methodology presented in this study as a practitioner-friendly, cost-effective monitoring scheme for peatland restoration. By using vegetation as a proxy, the sampling of near-surface peats and the application of assessment tools, robust information is collected to inform on central goals relating to peatland restoration.

4.4. Key words

Assessment, conservation, drainage, ecosystem functions, restoration success, soil, trophic condition, vegetation

4.5. Introduction

Near natural peat-accumulating peatlands, referred to as 'mires' (Joosten et al. 2017), and their biodiversity provide several ecosystem functions, such as nutrient and carbon storage, and landscape water regulation (Luthardt & Wichmann 2016). Large-scale drainage has reduced mires to only 10% of Europe's and less than 2% of Germany's former peatland area, leading to the loss of mire-specific biodiversity and ecosystem functions (Succow & Joosten 2001; Joosten & Tanneberger 2017). Whereas the majority of German peatlands has been drained for agriculture (72%), the second main reason for drainage is forestry (14%) (Trepel et al. 2017).

Pressure and efforts to restore peatlands have risen globally, as evidenced in numerous political policies and guidelines, such as the United Nations Environment Assembly (UNEA) resolution on 'Conservation and Sustainable Management of Peatlands' (UNEA 2019; Tanneberger et al. 2021). Ecological restoration is the 'process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed' (SER 2004), with restoration measures reducing the source of degradation. The most common measures in peatlands are hydraulic, e.g. blocking drainage systems. In addition, techniques such as extracting topsoil, removing non-native trees or reintroducing typical vegetation are applied (Gonzáles et al. 2014; Pfeifenberger & Fock 2015; Bonn et al. 2016). Peatland restoration aims at re-establishing ecological quality, natural structures and functions as well as biodiversity (Vasander et al. 2003; Bonnett et al. 2009; Aapala & Similä 2014). Specifically, a water table close to the surface (and thereby the reduction of greenhouse gas (GHG) emissions), the re-establishment of mire-specific biodiversity and, ultimately, the re-establishment of peatlands as carbon sinks are common goals of peatland restoration (Tanneberger et al. 2022).

To describe and evaluate change in respect to the set objective, and to learn from experience for future projects, monitoring is an essential part of restoration (Bonnett et al. 2009; Luthardt 2010). Studies in this regard on the effects of peatland restoration vary in their focus, ranging from restoration effects following peat extraction (Lucchese et al. 2010; Gonzáles et al. 2013; Rochefort et al. 2013) and agricultural use (Krejčová et al. 2021) to peatlands drained for forestry (Aapala & Tukia 2008; Haapalehto et al 2011; Laine et al.

2011). Some studies focus specifically on the restoration of fens (Mälson et al. 2008; Kreyling et al. 2021), bogs (Strobel et al. 2019) or *Sphagnum*-dominated peatlands (Kareksela et al. 2015). Monitored parameters include vegetation (Mälson et al. 2008; González et al. 2013), fauna (Duinen et al. 2002), hydrology (McCarter & Price 2013), soil and water properties (Armstrong et al. 2010; Ahmad et al. 2020, Hammerich et al. 2024) or the measurement (Komulainen et al. 1999; Wilson et al. 2016a) or estimation (Herrmann et al. 2018; Martens et al. 2022) of GHG emissions.

To date, these studies indicate that restoring peatlands to their natural state is a challenge. Whereas rising water tables and savings in GHG emissions seem to apply in the short term, mire-specific biodiversity or land cover characteristics are only, if at all, reached medium- or long-term (e.g. Aapala et al. 2014; Renou-Wilson et al. 2018; Kreyling et al. 2021). Furthermore, *Sphagnum*-dominated peatlands seem to re-gain natural peat growth rates in the surface layer (Kareksela et al. 2015; Hammerich et al. 2024), but depositing new peat after restoring non-moss peats appears to be complex and is only starting to be understood (Michaelis et al. 2020; Mrotzek et al. 2020).

Conditions likely to influence restoration success vary highly between sites, and the effects of restoration are still under investigation (e.g. Grand-Clement et al. 2013; Kreyling et al. 2021; Hammerich et al. 2024). An essential element for future restoration success involves collecting and publishing additional, long-term monitoring data on peatland restoration and learning from these results (Vasander et al. 2003; Haapaletho et al. 2011; Andersen et al. 2016).

The aim of the study presented herein is first to research the relations between restoration measures, groundwater abstracting facilities in the peatland catchment area, trophic conditions, acid-base ratios before restoration, hydrogenetic mire type, peatland size and years since restoration on restoration success. Second, we demonstrate the effects of peatland restoration in forests on the water table, peat accumulation, mire-specific biodiversity and GHG emissions. Third, based on our results, we aim to reflect on implications for future restoration projects. Finally, by using a cost-effective, comparably easy applicable monitoring programme, we try to set an example for other restoration projects. The study is situated in north-east Germany (federal state of Brandenburg).

4.6. Methods

4.6.1. Study area and study case

The study area (Brandenburg, north-east Germany, Figure 4.1) is characterised by different glacial advances, which have led to a high range of geomorphological formations and thereby favoured the development of different hydrogenetic and ecological mire types (Succow 1988; Succow & Joosten 2001; Kühn 2014). Succow (1988) describes eight hydrogenetic mire types (seven geogenous fen types and one ombrogenous bog type) based on their differences in hydrological conditions during formation. Furthermore, Succow (1988) describes ecological mire types based on the chemical quality of the water that feeds them, so oligotrophic, mesotrophic and eutrophic mires are distinguished in terms of nutrient availability, whilst acidic, sub-neutral and calcareous mires are classed in terms of base saturation.

The long-term annual mean temperature is 9.3 K (2002-2021), which is 1.2 K higher than the reference period (1961-1990). The south of Brandenburg is generally warmer due to rising continentalism. The long-term annual rainfall is 528 mm (period 2002-2021), which is a decrease by 59 mm to the former period (DWD 2019; DWD 2022). Peatland water supply in the area is dependent on the surrounding environment for inflowing water, as the climatic water balance is negative (Luthardt & Zeitz 2014).

In Brandenburg, about 30,000 ha (18%) of peatlands are situated in forests, about 22,000 ha of which are wooded and 8,000 woodless (Riek et al. 2014; LfU 2016). At least 90% of these peatlands are degraded, meaning a water deficit and, therefore, the often oxygen-induced degradation of near-surface peats (NSF 2007). Hasch et al. (2007) define forest peatlands as wooded or woodless peatlands, whose water balance is significantly influenced by their completely or predominantly wooded catchment area. Since 2004, the 'Protection programme for forest peatlands in Brandenburg (*Waldmoorschutzprogramm Brandenburg*)' has restored peatlands via hydraulic engineering and management measures within the peatland area and the above-ground catchment area (Table 4.1). The programme aims at raising the water table to enhance water retention and thereby benefit surrounding forests. Further restoration targets include protecting and re-establishing of mire-specific biodiversity, reinitiating peat accumulation as well as reducing GHG emissions (Müller 2014).

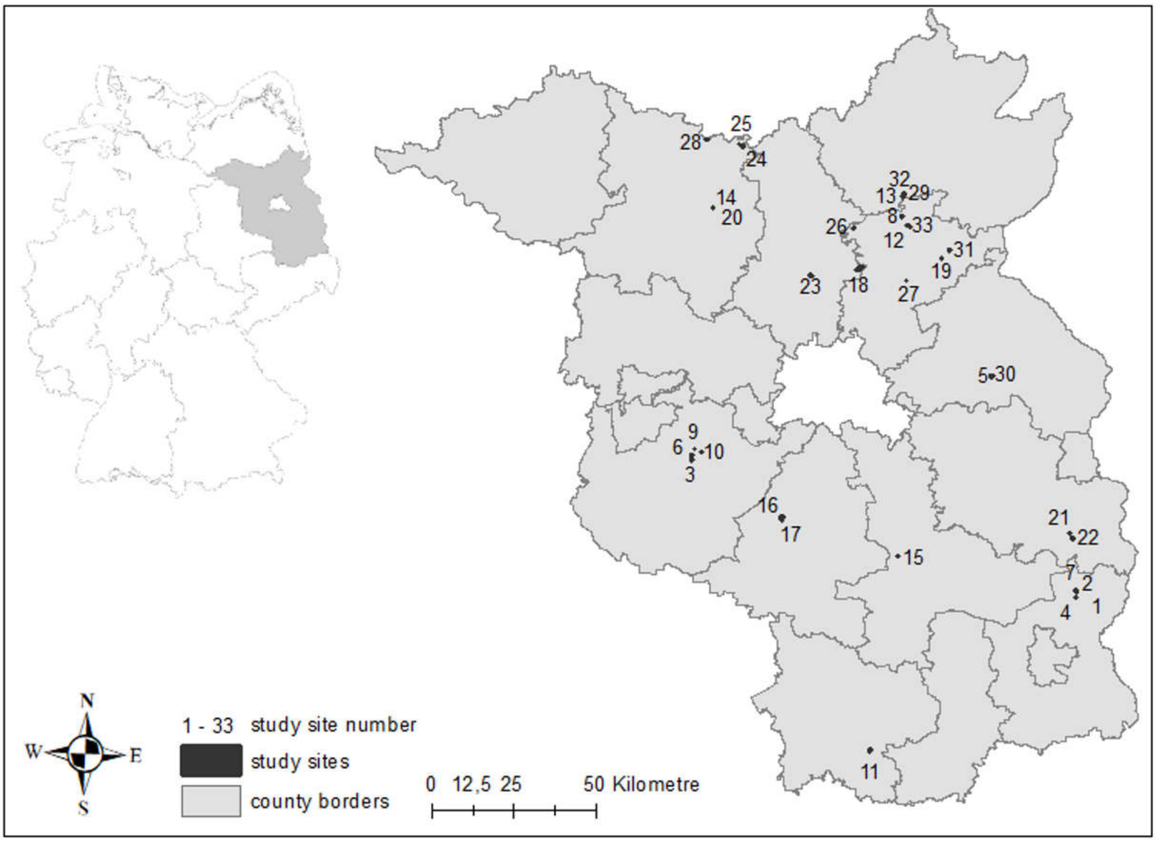


Figure 4.1: Location of the 33 study sites within the federal state of Brandenburg (north-east Germany).

We studied 33 restored peatlands (Figure 4.1) included in or associated with the 'Protection programme for forest peatlands in Brandenburg'. We chose the study sites based upon the (a) availability and quality of data before restoration, (b) representative distribution of hydrogenetic and ecological mire types (cf. Succow & Jeschke 1986) and location within the study area, (c) diversity of different degradation stages before restoration and (d) differences in restoration measures.

Table 4.1: Overview of restoration measures in the 'Protection programme for forest peatlands in Brandenburg (Waldmoorschutzprogramm Brandenburg)' (supplemented after LfU 2005; Müller 2014; MLUL 2016)

1. Hydraulic management	
1.1 Water retention	All hydraulic measures that hinder the outflow of water, such as blocking of ditches, removal of drainages or installation of sol sills.
2. Management measures	
2.1. Forest restructuring in the above-ground peatland catchment area	
2.1.1. Restructuring from pure stands of conifers to mixed stands with deciduous trees	Reduction of *Pinus sylvestris*-stocking (regardless of age), support (wildlife management, single-tree protection, light-favoured positioning) of upcoming deciduous trees or planting of deciduous trees to improve groundwater recharge in the immediate above-ground catchment area of the peatland.
2.1.2. Removal of spruces	*Picea* species and other non-site-adapted conifers located on the transition from peatland to the above-ground catchment area are removed, regardless of age.
2.2. Management measures within peatland area	
2.2.1. Removal of shrubs and trees	Partial and successive removal of spontaneously upcoming *Betula* species or *Pinus sylvestris* within the peatland area to reduce water consumption by trees and favour light-demanding plants.
2.2.2. Removal of tall vegetation	Mowing of tall sedges (e.g. *Carex acutiformis*) and reeds (e.g., *Phragmitis australis*) with the removal of mowed material to reduce nutrient availability and favour light-demanding plant species.

The distance between study sites ranged from less than 0.1 to more than 193 km. The study sites are fens, developed through terrestrialisation (n=10), percolation (n=9), water rise /paludification (n=7), kettle holes (n=6) and springs (n=1). Sixteen of the study sites are characterised as eutrophic, 14 as mesotrophic and three as oligotrophic. In 14 of the peatlands, acidic conditions dominate, and in 19 of the sites, subneutral conditions prevail. The study sites range in size from 0.2 to 42 ha with a median of 3.7 ha.

4.6.2. Field sampling

To map vegetation, we segregated areas within the peatland with homogenous physiognomic structures and floristic features. All units were then outlined on recent satellite images and, in unclear cases, with the use of GPS recordings. Data was transferred into ESRI ArcMap 10.5.1 to create spatial maps of each site. For each vegetation unit, all plant species were recorded with coverages, following Luthardt et al.

(2006). Furthermore, we sampled the upper 30 cm of topsoil peat, determining soil substrate, decomposition (according to Von Post 1924), colour and admixtures using Schulz et al. (2019).

4.6.3. Determination of the water table, peat accumulation, mire-specific biodiversity and GHG emissions

To transform the vegetation units into information on spatial water tables (soil moisture classes), trophic conditions and the acid-base-ratio, we applied the vegetation form concept created by Koska (2001), which uses vegetation as a proxy for on-site conditions.

To determine the areal water table for each vegetation unit, eco-sociological groups were derived, which indicate soil moisture classes (Table 4.2). To calculate one value to represent the whole peatland area, we addressed each soil moisture class by one calculation factor (Table 4.2), then multiplied the percentage share of each vegetation unit with the according calculation factor, added all values and then divided the sum by 100. For example, a value of 6 for the whole peatland area would mean a soil moisture class of 6+ (lower eulitoral) across 100% of the area. The most favourable would be a value of 5 and a soil moisture class of 5+ (wet) in 100% of the peatland area. However, this value could also be derived in different scenarios, e.g. when 50% of the peatland area had a soil moisture class of 4+ (very moist) and the other half had a soil moisture class of 6+.

The same calculation methodology was applied to calculate one value for determining changes in trophic conditions. The trophic condition groups and calculation factors are shown in Table 4.3.

We determined potential peat accumulation for each vegetation unit based on the uppermost peat layer, the water table and the presence of potentially peat-forming vegetation. We noted peat accumulation if the water table was close to the surface (soil moisture class of 5+), potentially peat-forming vegetation dominated and the peat showed slight to moderate decomposition (H1 to H7 according to Von Post 1924). Peat degradation was determined when oxygen-induced change from a natural peat structure to highly decomposed peat (H8 to H10 according to Von Post 1924), or a crumb structure ('earthified' peat) or even a fine granular structure ('murshified' peat) with a substantial change in soil properties, was observed (Ilnicki & Zeitz 2003; Schulz et al. 2019). New peat accumulation was noted when the water table was close to the surface and, additionally, a new layer of organic material in acidic peat moss-dominated peatlands on top of the degraded peat, or a recent root penetration within the degraded peat of peat-forming herbaceous plants, was visible (cf. Schulz et al. 2019; Hammerich et al. 2024).

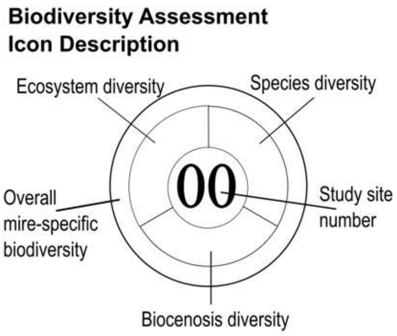
Table 4.2: Soil moisture classes according to the vegetation form concept (Joosten et al. 2015 after Koska 2001).

Soil moisture class	Verbal description	Water table relative to ground surface (+ above, - below)		Calculation factor
		Long-term median water table in the wet season	Long-term median water table in the dry season	
6+	Lower eulitoral	+150 to +10	+140 to 0 cm	6
5+	Wet	+10 to -5 cm	0 to -10 cm	5
4+	Very moist	-5 to -15 cm	-10 to -20 cm	4
3+	Moist	-15 to -35 cm	-20 to -45 cm	3
2+	Moderately moist	-35 to -70 cm	-45 to -85 cm	2

Table 4.3: Trophic level groups and trophic levels of peatland sites based on the carbon-based nitrogen content (Nc) of peat used in the vegetation form concept (Koska 2001; Succow & Stegmann 2001).

Trophic condition group	Trophic condition	Nc %	C/N ratio	Calculation factor
Oligotrophic (nutrient poor)	Very poor	< 2.5	> 40	1
	Poor	2.5 – 3.0	33 – 40	2
Mesotrophic (moderately nutrient poor)	Quite poor	3.0 – 3.8	26 – 33	3
	Moderate	3.8 – 4.9	20 – 26	4
Eutrophic (nutrient rich)	Strong	4.9 – 7.7	13 – 20	5
	Rich	7.7 – 10	10 – 13	6
Polytrophic (nutrient overloaded)	Very rich	10 - 13	7 – 10	7
	Extremely rich	> 13	< 7	8

Using the assessment tool for mire-specific biodiversity provided by Hammerich et al. (2022), we calculated values for each peatland site before and after restoration. The indicator-based tool assesses species level (number of mire-specific vascular plants and mosses), biocoenosis level (number of mire-specific and mire-typical plant formations and special habitats, as well as biotope connectivity) and ecosystem level (water table depth and near-surface peat conditions), scoring between 0 and 5 points for each factor. The values for each level are summed up to categorise a peatland's biodiversity, ranging from 'not mire-specific' (0 points) to 'very highly mire-specific' (15 points) (Figure 4.2).



Class	Accumulated value	Verbal description	Colour code
5	14-15	Very high mire-specific biodiversity	
4	11-13	High mire-specific biodiversity	
3	8-10	Moderate mire-specific biodiversity	
2	5-7	Low mire-specific biodiversity	
1	2-4	Very low mire-specific biodiversity	
0	0-1	no mire-specific biodiversity	

Figure 4.2: Overall assessment of mire-specific biodiversity based on the accumulated indicator values for species, biocoenosis and ecosystem levels, as well as an icon description for visualisation (Hammerich et al. 2022).

To estimate GHG emissions before and after restoration, we used the Gas-Emissions-Site-Type (GEST) model (Couwenberg et al. 2011; Spangenberg 2011; Reichelt 2015). By using vegetation as a proxy, which indicates long-term water table and nutrient status, which both affect GHG emissions, global warming potential (GWP) was estimated in t CO_2-eq. ha^{-1} $year^{-1}$.

To compare all factors for each peatland, before and after restoration, we checked and, if necessary, converted the data from before restoration for usability and applied the same assessment tools.

4.6.4. Statistical analysis

We performed a principal component analysis (PCA) to find patterns in the variables showing success (soil moisture class, gain in peat-accumulating area, gain in mire-specific biodiversity, lower GHG emissions) and possible influencing factors (restoration measures, groundwater abstracting facilities in the peatland catchment area, trophic condition before restoration, the acid-base-ratio before restoration, hydrogenetic mire type, peatland size and years since restoration). To describe changes in the water table, peat accumulation, mire-specific biodiversity and GHG emissions, as well as mire-specific species, plant formations and special habitats before and after restoration, we first tested all data for normality by using the Kolmogorov-Smirnov test. We tested for significant differences using paired t-tests for normally distributed samples, or a Wilcoxon signed-rank test for not normally distributed samples. Statistical significance was assumed if the p-value was < 0.05. All analyses were performed with R.

4.7. Results

Within the total sample (n=33, Table 4.4), all analysed components improved after restoration, meaning a rise in the water table, a gain in the peat-accumulating area, an increase in mire-specific biodiversity value and a reduction in GHG emissions (Table 4.5).

Table 4.5: Values for the soil moisture class, peat-accumulating area, mire-specific biodiversity and GHG emissions before and after restoration of the total sample (n=33). The p-value indicates whether there was a significant difference when comparing before and after the restoration measures.

	Soil moisture class		Peat accumulating area (% of total area)		Mire-specific biodiversity (0-15 points)		GHG emissions (t CO_2-eq. ha^{-1} $year^{-1)}$	
	before	after	before	after	before	after	before	after
Mean	3.96	4.40	26.84	55.53	8.97	10.86	10.51	7.39
Median	4.04	4.46	12.94	59.22	9	10	8.60	7.21
p-value	< 0.01		< 0.01		< 0.01		< 0.01	

Table 4.4: Overview of study sites with study site numbers (NO) and information on restoration measures (RMs), groundwater abstracting facilities in the peatland catchment area (GWA), dominant trophic condition group (TC) and the acid base ratio (ABR) before restoration, hydrogenetic mire type (HM), peatland size (SIZE), years since restoration (YSR) and soil moisture class, mire-specific biodiversity, peat accumulating area, GHG emissions and trophic conditions before (B) and after (A) restoration as well difference (BA). A grey background highlights an improvement in respect to restoration success, in this case a rise in the soil moisture class value, a higher mire-specific biodiversity value, a rise in the peat-accumulating area, lower GHG emissions and, additionally, lower nutrient availability.

NO	RM	GWA	TC	ABR	HM	SIZE	YSR	Soil moisture class			Mire-specific biodiversity (0 to 15 points)			Peat accumulating area (in % of total area)			GHG-emissions (t CO_2-eq. ha^{-1} $year^{-1}$)			Trophic condition		
								B	A	BA	B	A	BA	B	A	BA	B	A	BA	B	A	BA
1	1	2	2	2	2	2.5	12	4.2	3.2	-1	9	9	0	21.6	7.9	-13.6	14.7	11.6	-3.1	3.7	3.8	0.2
2	1	2	2	1	2	2.3	12	4	3.1	-0.9	7	6	-1	14.9	2.9	-12	8.6	12.5	3.9	4.0	4.2	0.2
3	1	1	3	1	2	3.6	13	4.3	3.7	-0.6	9	8	-1	31.2	28.0	-3.2	11.0	11.4	0.4	5.9	5.4	-0.6
4	1	1	1	2	1	3.7	12	4.8	4.4	-0.4	14	13	-1	81.1	41.6	-39.5	4.0	6.6	2.6	1.5	2.3	0.8
5	1	1	2	2	1	3.4	13	4.3	4.2	-0.1	13	14	1	61.9	60.3	-1.6	5.2	7.9	2.7	3.4	4.2	0.7
6	1	1	2	2	5	2.0	13	5	5	0	7	9	2	0.0	22.7	22.7	3.5	4.3	0.8	4.0	4.6	0.5
7	1	2	2	1	2	5.1	12	4	4	0	8	8	0	9.37	6.17	-3.2	8.2	8.1	-0.2	4.3	5.4	1.1
8	2	1	3	1	5	22.7	12	4.9	5	0.1	13	14	1	94.1	99.7	5.6	3.5	3.7	0.1	3.4	3.4	0.1
9	1	1	3	1	1	0.8	13	4.9	5	0.1	11	11	0	94.1	100	5.9	6.2	6.8	0.5	4.1	4.0	-0.1
10	1	1	1	2	5	1.3	13	3.9	4	0.1	5	5	0	0.0	3.6	3.6	12.8	12.5	-0.3	4.7	4.4	-0.3
11	2	1	1	2	4	23.3	14	4	4.2	0.2	10	13	3	31.6	60.2	28.6	8.2	7.9	-0.4	3.7	3.5	-0.2
12	2	1	2	2	4	2.0	8	2.8	3	0.2	4	5	1	0.0	0.0	0.0	22.0	16.4	-5.6	6.1	5.1	-1.0
13	2	1	2	2	5	3.0	12	5	5.2	0.2	10	12	2	50.4	57.0	6.6	4.2	3.3	-0.9	5.6	4.9	-0.7
14	1	1	2	2	1	1.5	12	4	4.2	0.2	11	12	1	0.0	23.9	23.9	7.5	6.3	-1.2	2.0	2.4	0.4
15	2	1	3	1	3	1.7	16	5	5.3	0.3	12	11	-1	96.3	75.4	-21.0	6.5	3.4	-3.1	6.0	4.6	-1.5
16	2	2	3	1	4	42.5	16	4.1	4.4	0.3	13	14	1	12.9	53.7	40.8	11.8	9.0	-2.7	3.7	3.4	-0.3
17	2	1	3	1	4	4.0	16	4.3	4.6	0.3	12	13	1	32.4	72.1	39.7	5.0	2.3	-2.6	2.9	3.2	0.3
18	2	1	2	1	4	34.9	4	3.2	3.6	0.4	7	8	1	0.0	21.8	21.8	16.4	14.3	-2.1	6.0	5.0	-1.0
19	2	1	2	2	4	9.6	27	4.7	5.1	0.4	15	15	0	77.4	88.0	10.7	7.8	4.5	-3.4	4.7	4.8	0.1
20	1	1	3	1	1	1.3	12	4	4.4	0.4	11	12	1	0.0	42.0	42.0	7.5	4.9	-2.6	2.0	2.6	0.6
21	2	1	3	1	2	0.7	12	4.2	4.7	0.5	8	10	2	20.2	65.0	44.8	7.8	6.6	-1.2	4.4	4.0	-0.4
22	2	1	2	1	2	5.6	10	4	4.5	0.5	10	11	1	19.8	69.2	49.5	11.6	7.2	-4.4	5.0	4.7	-0.3
23	2	1	3	1	5	12.1	6	3	3.6	0.6	5	8	3	0	1.33	1.33	16.4	10.8	-5.6	6.6	5.7	-0.9
24	2	1	4	1	4	10.6	13	4.2	4.8	0.6	12	14	2	10.4	59.2	48.8	8.5	7.4	-1.1	5.4	4.7	-0.6
25	2	1	3	1	4	2.2	12	4.2	5	0.8	13	14	1	39.7	100	60.3	12.8	2.8	-10	4.1	3.6	-0.5
26	2	1	2	2	2	6.1	19	3	3.9	0.9	4	6	2	0	29.1	29.12	19.5	7.7	-11.8	7.0	5.3	-1.7
27	1	1	4	1	1	1.5	10	4	4.9	0.9	6	9	3	0.0	95.0	95.0	7.5	3.3	-4.2	4.0	3.0	-1.0
28	2	1	3	1	2	7.2	19	3.5	4.5	1	6	10	4	0.0	51.5	51.5	15.3	10.2	-5.1	6.5	4.6	-1.9
29	2	1	3	1	5	1.1	13	3	4.3	1.3	5	9	4	0.0	18.5	18.5	9.3	7.9	-1.4	6.0	5.6	-0.4
30	2	1	3	1	2	4.3	13	3.3	4.9	1.6	6	9	3	0.0	29.2	29.2	9.9	3.0	-6.9	4.7	5.1	0.4
31	2	1	3	1	5	4.0	25	3	4.9	1.9	5	10	5	0.0	93.9	93.9	12.5	7.0	-5.5	6.0	5.5	-0.5
32	2	1	3	1	4	4.8	12	3.1	5	1.9	9	13	4	3.5	98.5	95.0	15.3	3.3	-12.1	4.8	4.3	-0.4
33	2	1	4	1	4	5.4	8	2.3	4.4	2.1	6	9	3	0.0	72.3	72.3	25.5	9.3	-16.2	6.6	5.1	-1.5

NO: Study site number RM: 1= Management; 2= Hydraulic GWA: 1= Not present; 2= present TC: 1= oligotrophic; 2= mesotrophic; 3= eutrophic	ABR: 1= subneutral; 2= acidic SIZE: 1= <2ha; 2= 2-4ha; 3= 4-6ha; 4= >6ha HM: 1= Kettle hole mire; 2= Percolation mire; 3= Spring mire; 4= Terrestrialisation mire; 5= Water rise mire

Results of the PCA are shown in Figure 4.3. The proportion of variance explained by PC 1 is 39.3% and for PC 2 15.6%, amounting to a cumulative proportion of 54.9% on the first two axes. The PCA indicates a positive correlation between gains in the soil moisture class, in the peat formation area and in mire-specific biodiversity, along with the negative correlation of these with GHG emissions from before to after restoration. Variables describing restoration success (gains in the soil moisture class, the peat-accumulating area, mire-specific biodiversity and lower GHG emissions) are not correlated with the acid-base ratio and trophic conditions before restoration, hydrogenetic mire type, peatland size and change in trophic conditions from before to after restoration, but they are positively correlated with years since restoration. Peatlands undergoing management measures were distinguishable from peatland undergoing hydraulic measures in terms of restoration success. Whereas kettle hole mires were only represented in peatlands undergoing management measures, terrestrialisation and water rise mires were mainly represented in study sites undergoing hydraulic measures. We also found a significant negative effect, i.e., a loss of 0.5 points in mire-specific biodiversity (Figure 4.4), where groundwater-abstracting facilities were present in the peatland catchment area (n=4).

Restoration success was significantly higher where hydraulic measures were applied. Therefore, we divided the sample into peatlands undergoing hydraulic measures and peatlands undergoing management measures. We excluded all peatlands (n=4) where groundwater-abstracting facilities were present in the catchment area because we argue that the applied restoration measure did not have the highest impact on restoration success but groundwater abstraction did in this regard. In this analysis, peatlands undergoing management measures did not show significant changes in any of the analysed components (Figure 4.5).

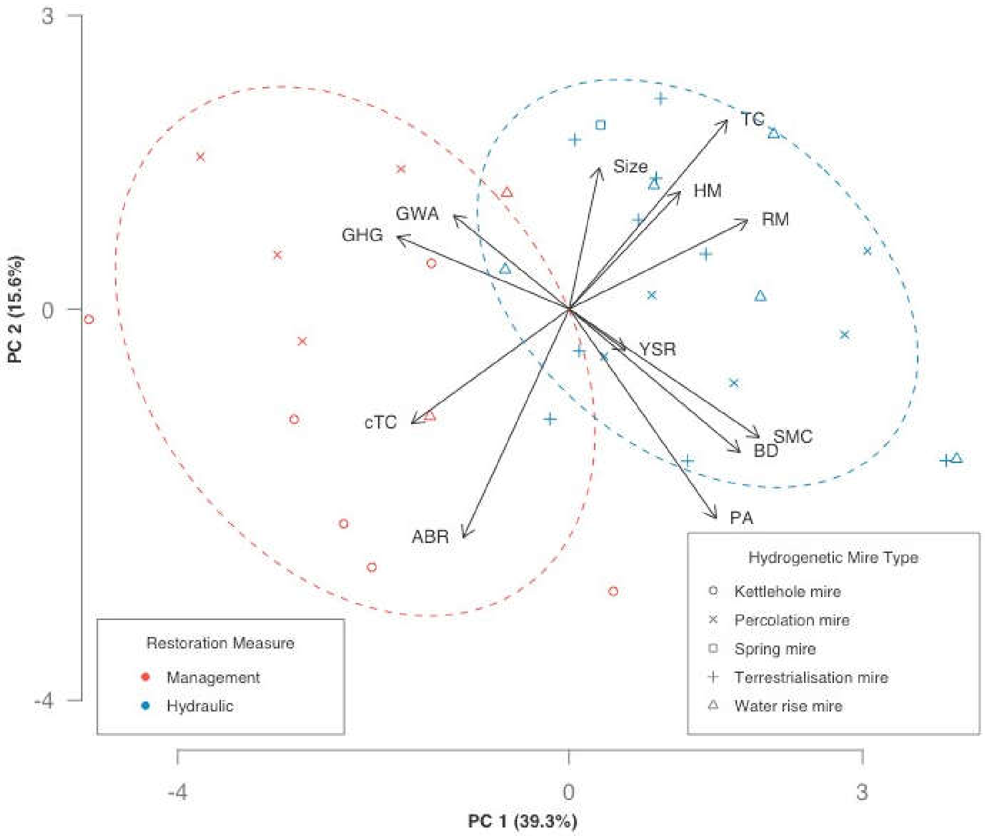


Figure 4.3: Principal component analysis showing the variables gain/loss in soil moisture class (SMC), peat accumulating area (PA), mire-specific biodiversity (BD) and GHG emissions which describe restoration success. Further the associated factors of the acid base ratio before restoration (ABR), trophic conditions before restoration (TC), hydrogenetic mire type (HM), years since restoration (YSR), peatland size (size) and changes in trophic conditions from before to after restoration (cTC) are shown.

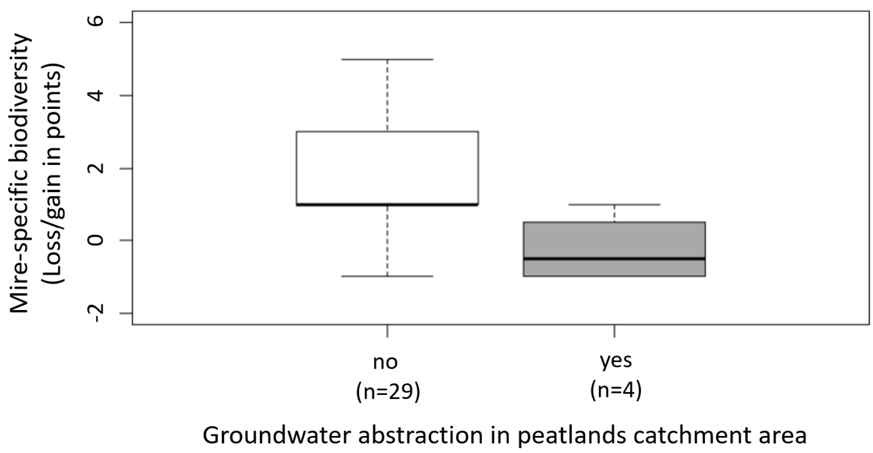


Figure 4.4: Impact of groundwater-abstracting facilities in the peatland catchment area in terms of gains in mire-specific biodiversity after restoration.

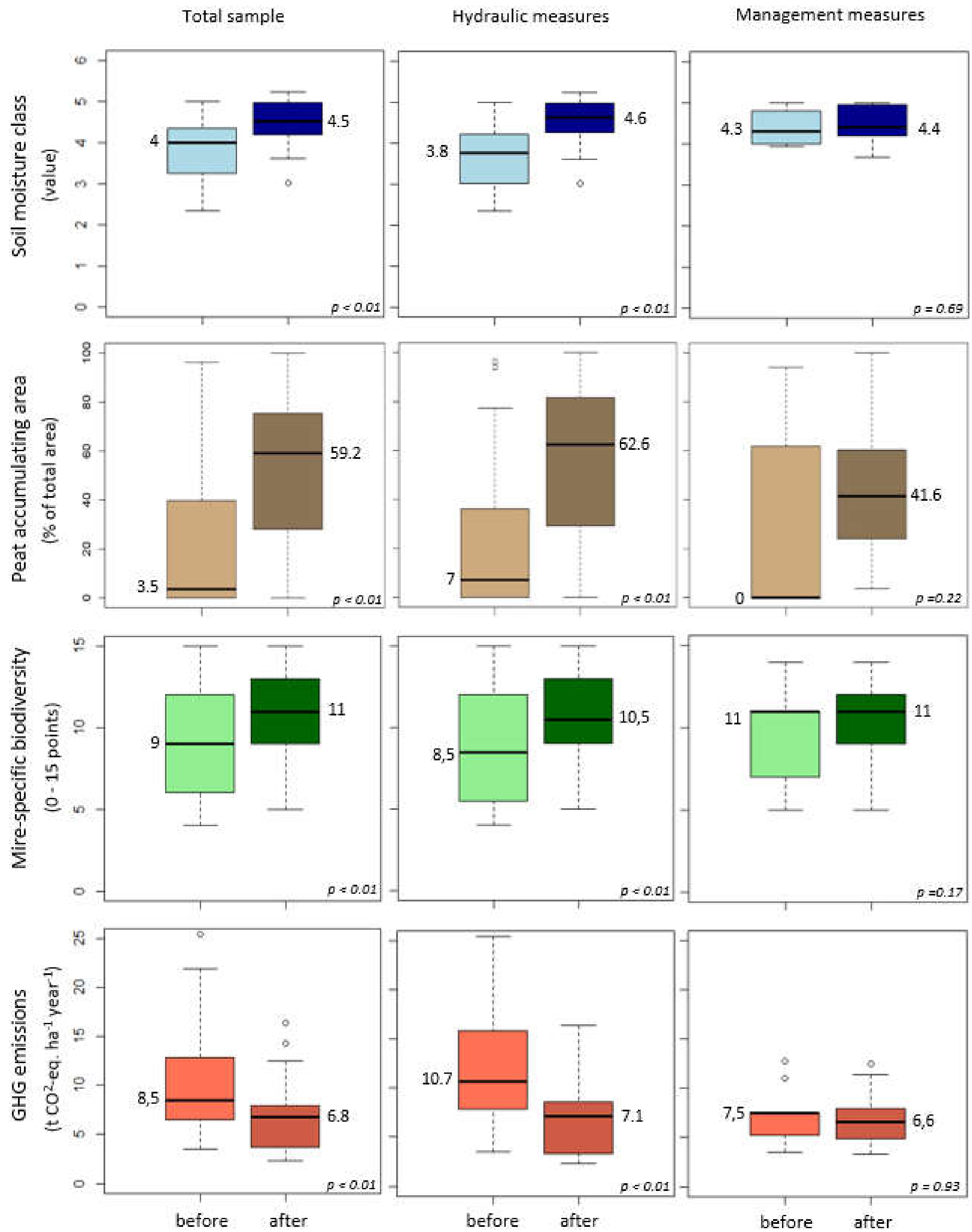


Figure 4.5: Change in the soil moisture class, peat accumulating area, mire-specific biodiversity and GHG emissions before and after restoration, shown for the reduced sample (n=29, study sites with groundwater abstracting facilities in the peatland catchment area were excluded), for study sites which have undergone hydraulic measures (n=20) and for the study sites which have undergone management measures (n=9).

The soil moisture class within the total sample increased from before to after restoration from a median of 4 to 4.5. Peatlands restored by hydraulic measures improved significantly from a median of 3.8 before to 4.6 after restoration. Peatland undergoing management measures showed a median of 4.3 before and 4.4 after restoration, but the slightly higher values were not significantly different.

The percentage of peat-accumulating area within the total sample increased from before to after restoration from a median of 3.5% to 59.2%. The peatlands restored by hydraulic measures improved significantly from a median of 7% before to 62.6% after restoration. Peatlands undergoing management measures showed a median of 0% before and 41.6% after restoration, yet the higher values were not significantly different.

Mire-specific biodiversity within the total sample improved from before to after restoration from a median of 9 to 11 points. Peatland restored by hydraulic measures improved significantly from a median of 8.5 before to 10.5 points after restoration. Peatlands undergoing management measures showed no change with a median of 11 points before and after restoration. Furthermore, the number of mire-typical vascular plants, plant formations and special habitats increased significantly after rewetting. In contrast, the slight increase in the highly specialised mire-specific vascular plants was not significant (Table 4.6).

Table 4.6: Comparison of mire-typical and mire-specific vascular plants, plant formations and special habitats before and after restoration measures were initiated. The p-value indicates whether there was a significant difference when comparing before and after the restoration measures.

	Mire-typical vascular plants		Mire-specific vascular plants		Plant formations		Special habitats	
	Before	After	Before	After	Before	After	Before	After
Mean	12.4	22	5	5.7	2	2.88	3.12	4.12
Median	8.5	20,5	2	5,5	2	3	3	4
p-value	< 0.01		= 0.15		< 0.01		< 0.01	

GHG emissions estimated via GEST approach within the total sample decreased from before to after restoration from a median of 8.5 to 6.8 t CO_2-eq. ha^{-1} $year^{-1}$. Peatlands restored by hydraulic measures showed a significant decrease from a median of 10.7 before to 7.1 t CO_2-eq. ha^{-1} $year^{-1}$ after restoration. Peatlands undergoing management measures showed a median of 7.5 before and 6.6 t CO_2-eq. ha^{-1} $year^{-1}$ after restoration, but the slightly lower values were not significantly different.

A visual example of development after hydraulic in comparison to management measures for all researched parameters is given in Figure 4.6.

4.8. Discussion

4.8.1. Restoration outcome

Herein, we demonstrate that peatland restoration generally leads to higher water tables, an increase in the peat-accumulating area, higher mire-specific biodiversity and lower GHG emissions. However, our results also indicate that although positive effects can be achieved, peatlands are not returned comparably to near-natural conditions within the studied time frame of 27 years.

Of our studied peatlands, 26 out of 33 showed, even if just slightly, higher water tables after restoration. Only eight were wet with a soil moisture class 5+ or higher in at least 90% of the peatland area (Table 4.4). Although restoring peatland hydrology and raising the water table closer to the surface is a cornerstone in successful restoration, getting peatlands wet seems to be a challenge. The main reason is high anthropogenic alteration, which leads to total changes in vegetation, a lower water table in surrounding lands or peat subsidence – and therefore a different surface structure (Convention on Wetlands 2021). Comparable studies show that the water table can often be raised by rewetting efforts (Haapalehto et al. 2011; Laine et al. 2011; Luthardt et al. 2021), but outcomes differ in comparison to pristine sites (Krejčová et al. 2021; Kreyling et al. 2021).

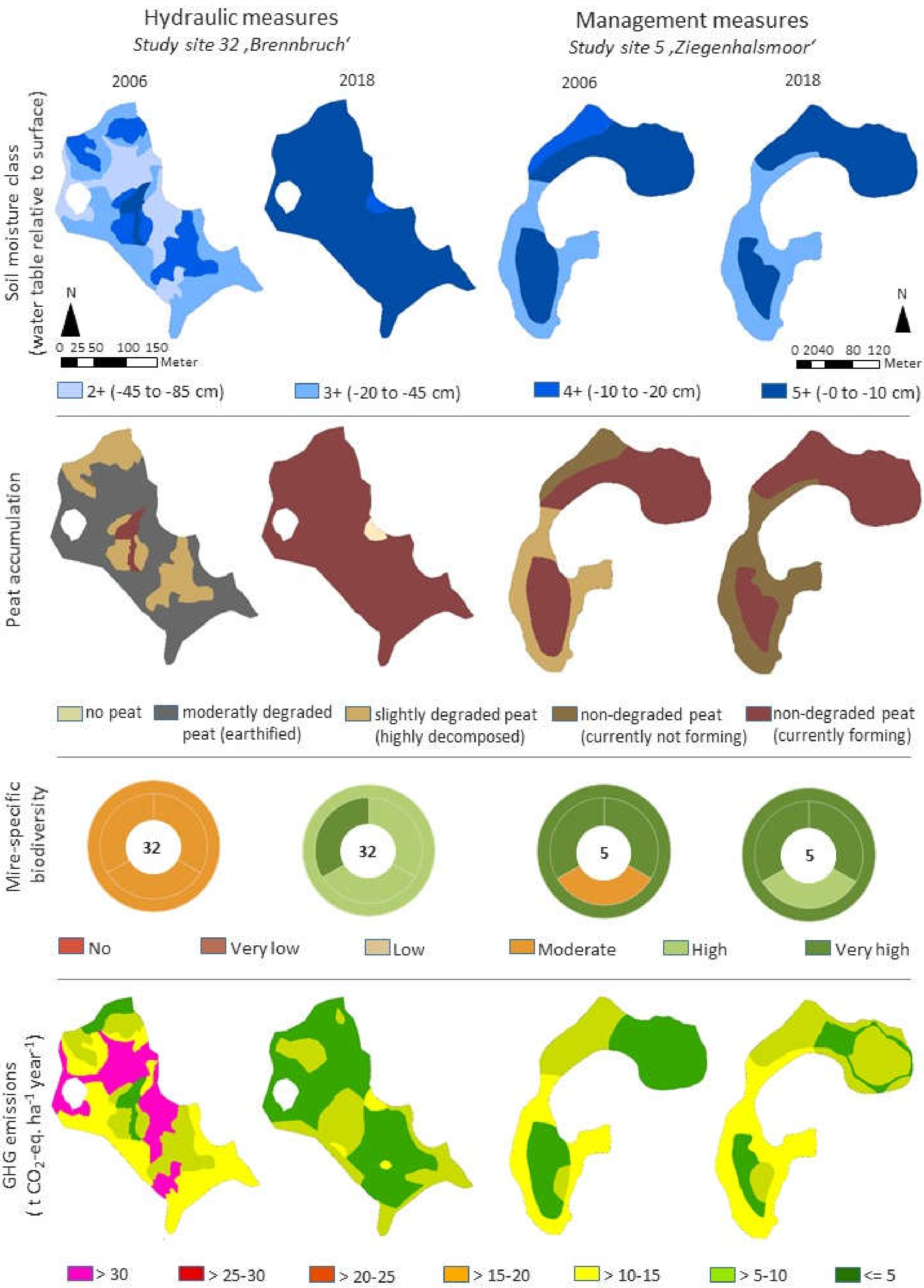


Figure 4.6: Changes in the soil moisture class, peat degradation stage, mire-specific biodiversity and GHG emissions before and 12 years after hydraulic measures or management measures were initiated are shown for two examples. Brennbruch (study site 32) was restored by hydraulic measures, while Ziegenhalsmoor (study site 5) has undergone management measures, as no drainage ditches were installed but signs indicating a water deficit were present. Whereas Brennbruch showed a high rise in the water table, the peat accumulating area, mire-specific

biodiversity and savings in GHG emissions, Ziegenhalsmoor was in a better initial state when management measures were initiated, and it remained relatively stable over the 12-year period.

First, water table amplitudes in rewetted fens were found to be larger than in pristine sites, most likely due to the loss of a functioning 'acrotelm' (Tolonen & Turunen 1996), namely the highly dynamic near-surface peat layer, which actively takes up water. The mineralisation and subsidence of the near-surface peat layer following drainage results in lower organic matter and higher bulk density, which in turn results in decreased hydraulic conductivity and decreased porosity (Convention on Wetlands 2021; Kreyling et al. 2021). Second, rewetting affects areas within the peatland differently, as drainage results in often strong relief differences. Most commonly, areas around ditches are lower than their surroundings, due to high peat subsidence through effective drainage. After rewetting, water flows into these lower areas, which leads to drier sites further away from the ditch, especially the higher they are compared to other areas within the peatland (e.g. Haapalehto et al. 2011; Hedberg et al. 2012; Strobel et al. 2019). Third, vegetation influences water table dynamics. The vegetation of mires is characterised by great water storage capability, e.g. peat mosses, which can take up around 20% of their dry weight in water (Rice 2009). In this context, we observed in study sites 25, 32 and 33 near-natural, often central areas remaining with peat-forming mire-typical or mire-specific vegetation (e.g. peat mosses, tall sedges), successful rewetting with the quick re-colonisation of higher degraded areas (Figure 4.7) and an additional new acrotelm formation. This is in line with comparable literature, as Grootjans & van Diggelen (1995) describe that the most effective, short-term restoration should focus on the least degraded peat systems.

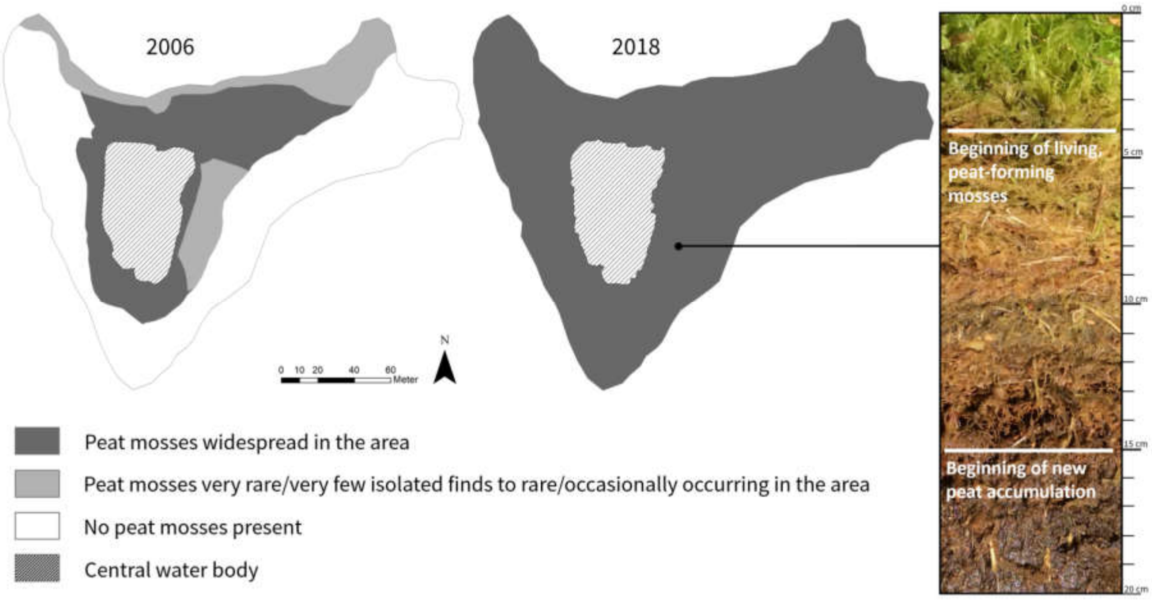


Figure 4.7: Comparison of peat moss (Sphagnum sp.) occurrence in Teufelsbruch (study site 25) before (2006) and 12 years after restoration (2018). Peat mosses have grown over the degraded, highly decomposed peat and initiated new peat accumulation. Abundance based on Luthardt et al. (2006).

Peat accumulating areas in our study sites increased by 46%. Especially peat moss-dominated peatlands seem to quickly start accumulating dead organic matter, up to 1.5 cm/y (study site 25). Hammerich et al. (2024) found new peat moss peat layers with a mean thickness of 6.8 cm to 36.1 cm, with mean accumulation rates of 0.38–2.01 cm/y in restored fens within 17 years after restoration. Kareksela et al. (2015) found that within 5 and 10 years after restoration, the surface layer growth of restored peatlands was similar to pristine sites of *Sphagnum*-dominated peatlands, but the carbon sequestration rate within the surface layer was lower. Hammerich et al. (2024) describe a general gain in total organic carbon and a decrease in dry bulk density in near-surface peat after restoration, citing it as one key factor in re-establishing peatlands as carbon sinks. In respect to sedge-dominated peatlands, Mrotzek et al. (2020) and Michaelis et al. (2020) determine a ‘proto-peat’ consisting of roots, radicals and litter 20 years following restoration. Furthermore, next to new peat accumulation on top of older, degraded peat, a structural change in the former degraded peat, showing larger re-aggregates in a sludge mass and actively in-growing roots forming displacement peat, has been described (Hammerich et al. 2024). In our study sites 24 and 33, where sedges re-colonialised degraded earthified or murshified peat soils, the latter was also evident. In conclusion, our – and comparable – findings indicate that restoration can positively influence key processes which lead to the re-establishment of the peatland function to accumulate peat and store carbon. However, as near-surface peat is highly dynamic, due to a fluctuating water table, the newly added material will undergo continuous decomposition, and the amount of carbon transferred into long-term storage remains unclear (Young et al. 2019). This is especially important in respect to climatic change. In the context of reducing GHG emissions, far more important than restoring the carbon sink function is the conservation of the already accumulated peat body (Convention on Wetlands 2021).

Our results show that through peatland restoration, mire-specific biodiversity is enhanced, albeit only slightly. Only six out of 33 study sites are highly mire-specific after restoration, which is in line with the literature stating, that peatland restoration increases biodiversity, although it is not comparable to near-natural conditions (e.g., Hedberg et al. 2012; Krejčová et al. 2021; Kreyling et al. 2021). Study sites with a very high value (study sites 5, 8, 16, 19, 24 and 25), which indicates near-natural conditions, showed high to very high mire-specific biodiversity before restoration. Highly degraded sites (very low to moderate value) showed the highest rises in mire-specific biodiversity (28, 29, 31, 32) of 4 to 5 points, but only study site 32 of this cohort achieved a high biodiversity value after restoration. The water table in particular can be raised in the short term, leading to peat accumulating conditions, higher plant formation heterogeneity in the midterm and a slow rise in mire-specific species. It is important to state that in comparable studies, biodiversity often refers to vegetation or fauna, whereas Hammerich et al. (2022) describe mire-specific biodiversity based on mire-specific species, typical structures, biotope connectivity, water table depth and peat formation. Vegetation recovery towards pristine conditions seems slow after restoration (Mälson et al. 2008). Strobel et al. (2019) found that whereas mire-specific plant species cover increased within an 18-year period after restoration, plant species number did not do so. Plant community composition was observed similarly when comparing drained peatlands after five years of restoration, whereas plant community composition of 10 years following restoration moved closer to comparable pristine sites (Karksela et al. 2015). Kreyling et al. (2021) describe for 222 rewetted fens a shift towards tall helophytes after rewetting as one main difference in plant community composition compared to near-natural sites. Within our dataset, mire-specific vascular plants only increased slightly in number, although not significantly (Table 4.6), but we observed a generally wider distribution after a favourable water table set in. The shift to vegetation with lower nutrient requirements (see Table 4.4) in most of the restored peatlands might positively influence the diversity of mire-specific species, which are often adapted to extreme nutrient conditions and pH values (cf. Minayeva et al. 2008). Furthermore, the number of plant formations and special habitats increased significantly after rewetting (Table 4.6), leading to a mosaic of vegetation formations and a more diverse habitat.

By using the GEST approach, we estimated a reduction of GHG emissions in 26 of the 33 studied peatlands at between 0.15 and 16.19 t CO_2-eq. ha^{-1} $year^{-1}$. In the remaining seven peatlands GHG emissions were estimated to be slightly higher, ranging from 0.13 to 3.87 t CO_2-eq. ha^{-1} $year^{-1}$. This is coherent with no apparent water table rise (n=3) or an even deeper water table (n=4) as well as a change in vegetation, especially the higher occurrence of shunt species, i.e., plant species with coarse aerenchyma that provide bypasses for methane fluxes (Couwenberg et al. 2011). All study sites remain contributors of GHG emissions with values of 2.32 to 16.27 t CO_2-eq. ha^{-1} $year^{-1}$. This is comparable to the current literature, stating that in general rewetted peatlands emit more CO_2-eq. than pristine sites, but savings compared to their drained stages are still high (Couwenberg et al. 2011; Wilson et al. 2016; Humpenöder et al. 2020). The IPCC calculates a reduction of 6 t CO_2-eq. ha^{-1} $year^{-1}$ for previously forested land use and 20 t CO_2-eq. ha^{-1} $year^{-1}$ for previous grassland use in the temperate zone (Barthelmes et al. 2015). Wilson et al. (2016), who include additional values post-2013, calculate an emission reduction of -0.32 t CO_2-eq. ha^{-1} $year^{-1}$ (forest land, nutrient-rich) to 8.47 t CO_2-eq. ha^{-1} $year^{-1}$ (forest land, nutrient-poor) and from 4.68 t CO_2-eq. ha^{-1} $year^{-1}$ (grassland, nutrient-rich, shallow drained) to 20.84 t CO_2-eq. ha^{-1} $year^{-1}$ (grassland, nutrient-poor). The slightly lower estimates in our study can be explained by a comparably low degree of drainage, with only one study site (n=33) being drained mostly -45 to -85 cm in the long-term median water table in the dry season. Peatlands in forests are usually characterised by less intensive use than peatlands under agriculture, often due to their smaller size and difficult accessibility. GEST types for wooded vegetation units (Spangenberg 2011) are interpolated from woodless GEST types and therefore present some uncertainties. Furthermore, GEST types for open, woodless vegetation units are relatively old from 2011 and 2015 (Couwenberg et al. 2011, updated by Reichelt 2015), and additional measurements since then are currently included in a revision of GEST values. We did not account for potentially high methane emissions directly after rewetting, as we compared the pre-drainage stage to a stage several years after restoration measures were initiated, not a timeline. However, some studies reveal high remaining methane emissions up to 10 years after restoration (Antonijević et al. 2023), while other studies argue that although methane emissions peak within the first few years, the climate change mitigation potential of peatland rewetting is not undermined (Couwenberg et al. 2011; Günther et al. 2020).

As restoring peatlands to functioning mires is evidently challenging, the necessity to conserve all remaining mires is clear. Still, even if a peat-accumulating, highly biodiverse state cannot always be restored, any improvement in the peatland's state is worth a try, as preservating the peat body and favouring wetland plants are important rewetting outcomes. Furthermore, the ecosystem function as a water storage basin, within the context of climatic change with increasing heavy rainfall events and a shift towards precipitation in winter (DWD 2019), is an important function to positively influence and regulate landscape water balance.

4.8.2. Hydraulic measures, management measures and other factors influencing restoration success

Our results reveal that positive restoration effects are mainly connected to hydraulic measures. Peatlands in our study drained by ditches were often in a worse condition before restoration and therefore have greater potential for improvement. On the contrary, the peatlands in our study undergoing management measures were still in a good overall conservation condition before measures were initiated, indicated by the median value of 11 points of mire-specific biodiversity. Commonly, no drainage ditch was installed, but the peatlands were characterised by a visible water deficiency, such as heavy tree growth or the colonisation of *Molinia* grass. The removal of trees within the peatland, forest restructuring in the above ground peatland catchment area and mowing were usually the only feasible management options. Although management measures did not lead to any significant improvement in the peatland's condition, they nevertheless preserved the pre-measurement conditions of the peatlands during the period under review. In the relevant literature, management strategies are assumed to enhance restoration success. Mowing, for example, can successfully restore species-rich fen vegetation in hydrologically well-preserved fen meadows (Grootjans & van Diggelen 1995). Mälson et al. (2008) argue that to maintain or support mire-specific vegetation, management intervention, such as the continuous removal of dominant trees, shrubs and tall vegetation, might be necessary. In the context of our study, however, climatic changes and reduced water supply in recent decades must also be considered, as they might have counteracted possible positive effects. In Brandenburg, mean annual temperature has risen by 1.3 K, with the driest year in 2018 since recording in 1881 (DWD 2019). We estimate that especially forest restructuring in the above-ground peatland catchment area is the only long-term option to positively influence peatlands, even though only slow positive effects can be expected due to the time required for changes to mixed or pure stands of deciduous trees.

Groundwater-abstracting facilities within the peatland catchment area had a negative effect on restoration success. Grootjans & van Diggelen (1995) state that due to intensive land-use practices and large-scale groundwater abstraction, the water discharge necessary for sustaining or reinitiating peat formation in fens is not sufficient.

In our study, years since restoration positively indicated restoration success, which is in line with the literature, stating that time since restoration has a significant positive influence on, for example, plant diversity or mire-typical vegetation (Karksela et al. 2015; Strobel et al. 2019). The variables trophic condition and acid-base-ratio before restoration, hydrogenetic mire type and peatland size were not correlated with restoration success. Nonetheless, in respect to hydrogenetic mire type, kettle hole mires in our study were commonly under management measures, not drained by ditches and in a better initial state than other hydrogenetic mire types. Possible explanations are the often difficult accessibility due to the high relief surrounding kettle hole mires and their usually small size. Heikkilä & Lindholm (1995) argue that mires in areas with a positive climatic water balance are easier to restore, because water is generally available. Furthermore, central parts of large mires are more likely improved by restoration, because they are not at all or less influenced by their surroundings; moreover, especially small peatlands (here bogs) with tree coverage and missing hollows might be very difficult to restore, or even not at all. Twenty-four out of the 33 studied peatlands are smaller than 6 ha, and they have a high dependency on the surrounding catchment area (cf. Luthardt & Zeitz 2014). The climatic water balance in the state of Brandenburg is negative. Therefore, to fully rewet peatlands in the long term, restoration needs to emphasise the peatland catchment area.

4.8.3. Monitoring approach

Monitoring of peatland restoration is not yet standardised and in particular faces the challenge of short timeframes, often related to project duration and funding (Bonnett et al. 2009; Andersen et al. 2016). Currently comparable monitoring studies are commonly based on vegetation plots or transects (e.g. Kareksela et al. 2015, Strobel et al. 2019, Krejčová et al. 2021). GHG emission factors are normally based on CO_2 flux measurements taken by chamber techniques or the eddy covariance method (Poyda et al. 2017). We recommend the methodological approach applied herein as a user-friendly, efficient and robust monitoring programme to evaluate peatland restoration success. One precondition is areal vegetation mapping with the additional sampling of near-surface peats in each vegetation unit before and at any time after restoration measures were initiated. We argue that by using vegetation as a proxy for on-site conditions, which reflect water table depth over several years and do not react directly to weather extremes such as draughts, robust information on the current state of the peatland can be derived. By analysing the entire area of the peatland, a holistic picture is created in contrast to single vegetation plots. Furthermore, the areal vegetation mapping can be used to apply the GEST approach for estimating GHG emissions and with small additions of special habitats and evaluating near-surface peat degradation the indicator system for mire-specific biodiversity can be applied.

4.9. Acknowledgements

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4.10. Author contributions

JH, JZ and VL designed the study. JH screened and reprocessed the data before restoration. JH collected field data after restoration from 2018 to 2020. JH and HvW compiled the statistical analyses. JH wrote the manuscript. CS, HvW, JZ and VL reviewed the manuscript. All authors contributed to the final version.

5. Synoptic Discussion

5.1. Choice of monitoring parameters

In this thesis, the water table, peat properties (pH value, TOC, N, C/N ratio, BD, peat type and structure), peat accumulation, mire-specific biodiversity and GHG emissions were the areas chosen for monitoring and assessing the success of peatland restoration. This choice of monitoring parameters is supported by various publications, in which one or more of them has been selected to describe the effects of peatland restoration projects (e.g., Hedberg et al. 2012; Kareksela et al. 2015; Herrmann et al. 2018). Van Bellen & Larivière (2020) reviewed peatland research literature from 1991 and found that the nine most researched topics were carbon, botany, carbon dioxide, methane, climate change, Holocene, nitrogen, biodiversity and hydrology (in order of number of publications), thereby proving that the parameters chosen herein reflect the most common research topics in current peatland science. A comprehensive, consensus-based study on what to monitor in peatland restoration is presented by Reed et al. (2022). Core areas with broad domains are described, namely climate (peatland condition, peat accumulation, GHG flux, water quality, fire damage, erosion), hydrology (topography, hydrological connectivity, water balance, groundwater flow, water table, surface water, moisture/water content) and biodiversity (habitat, vegetation, birds, invertebrates). For these broad domains, contextual variables are defined for monitoring purposes.

I argue that the chosen monitoring parameters of this thesis are the fundamental requirements for monitoring peatland restoration effects, as they reflect the central ecosystem components and functions of mires (cf. Parish et al. 2008) and are most commonly used in describing change after peatland restoration. In equal measure, they represent the restoration goals of the two studied restoration projects: The 'Protection programme for forest peatlands in Brandenburg' and peatland restoration projects with the EU-Life project 'Restoration of clear water lakes, mires and swamp forests of the Lake Stechlin' (Luthardt et al. 2002; Luthardt et al. 2021; Müller 2014). However, based on project goals, funding and specific research questions, I suggest integrating additional monitoring parameters (see e.g., Strobel et al. 2019; Tanneberger et al. n.d.). A good basis on which to choose them, depending on the research focus or the objective of a restoration project, is provided by Reed et al. (2022).

5.2. Assessment methodology

The research question addressed in the following chapter:

III a Which easy applicable monitoring methodology can be used to assess changes in the water table, peat-accumulating area, mire-specific biodiversity and GHG emissions, before and after peatland restoration?

Generally, the assessment methodology developed and applied in this thesis uses the areal mapping of vegetation in vegetation units, which are homogenous in floristic features and physiognomic structures (Koska 2001; Luthardt et al. 2006). Although, due to the persistence of species or slow immigration pathways, vegetation reacts slowly to change, such as changes in water table depth or nutrient availability (cf. Kareksela et al. 2015), the advantage over sampling plots in specific areas or along transects is the area-wide statement (cf. Luthardt et al. 2006). Additionally, the upper 30 cm of near-surface peat is sampled in each vegetation unit to determine the soil substrate, decomposition (according to Von Post 1924), colour and admixtures, using Schulz et al. (2019). This field data is then applied through different tools to evaluate changes before and after restoration. While discussing the assessment methodology, remarks and suggestions for transferability are made.

Water table

The vegetation typology in this thesis is based on the "vegetation form" concept developed by Koska (2001), which uses vegetation as a proxy to indicate on-site conditions. Consequently, soil moisture classes for wet and dry seasons can be determined for each vegetation unit, representing long-term median values for water table depth in relation to surface. The vegetation form concept is a widely accepted methodology in the study region (Succow & Joosten 2001; Couwenberg et al. 2015; Luthardt et al. 2021). Another commonly used tool in research to describe changes in the water table is water gauge measurement (e.g., in Ahmad et al. 2020; Strobel et al. 2019). Although continuous data can be provided, only the area where the water gauge is installed is represented, which is often central, near the (former) ditch and known to be wetter than marginal areas (cf. Hedberg et al. 2012; Strobel et al. 2019). Nevertheless, measuring water table depth in the first years following restoration can be helpful, when vegetation has not yet adapted to the changed conditions.

The vegetation typology used for his study, developed especially for north-east Germany, needs to be adapted for other biogeographical and climatic zones. Water gauges are recommended if similar vegetation typologies are unavailable or adaptable. To represent water table depth for the whole peatland, more than one water gauge might be necessary, and interpolation may also be required.

Peat properties and accumulation

In the six Stechlinsee peatlands within the current research, we analysed the peat properties pH value, TOC, N, C/N ratio, BD, peat type and structure. TOC values in successfully rewetted fens are comparable to values of undisturbed peats. Simultaneously, a lower BD was determined, and new peat accumulation was estimated in the form of new peat layers and displacement peat. Therefore, it is indicated that with a favourable water table and the dominance of peat-forming vegetation, peat accumulation can be re-established. These findings were used to estimate peat accumulation in the uppermost peat layer for each vegetation unit in the 33 'Protection programme for forest peatlands in Brandenburg' study sites. When the near-surface peat showed signs of previous drainage, and therefore peat degradation (high decomposition, earthification, murshification), new peat accumulation was determined if the water table was close to the surface, peat-forming vegetation prevailed and a new layer of organic material in acidic peat moss dominated or recent root penetration in sedge-dominated peatlands was determined. For undisturbed peats, peat accumulation was determined when the water table was close to the surface, peat-forming vegetation prevailed, and near-surface peat decomposition was slight to moderate (H1 to H7 according to Von Post (1924)).

As the process of peat accumulation is comparable worldwide, good transferability is expected; however, the methodological approach presented herein can only be used to describe the potential for new material being transferred into long-term C storage, as the highly dynamic near-surface peat will undergo continuous decomposition (see also Chapter 5.3 on restoration effects on peat accumulation; Young et al. (2019)).

Mire-specific biodiversity

The research questions addressed in the following section:

I a	Which indicators, attributes and measure values are suitable for assessing mire-specific biodiversity on different levels?
I b	Which value scales for each indicator represent the degree (none to very high) of mire-specific biodiversity?
I c	Is the assessment via indicators comparable to the assessment by experts and practitioners in peatland restoration?

A user-friendly, easy applicable indicator system was developed to assess mire-specific biodiversity, as existing assessment tools were insufficient or focused only on the species level of mire biodiversity. In contrast, the developed indicator system equally rates the levels of species, biocoenosis and ecosystem diversity. The integration of genetic diversity requires significant expert knowledge, and genetic research in the study area was found to be insufficient for usage in the indicator system.

Ranging from 0 (no mire-specific biodiversity) to 15 (very high mire biodiversity) points for the overall evaluation, mire-specific biodiversity was assessed against three indicators for the three levels of biodiversity, each rated with up to 5 points. The number of mire-specific vascular plants and mosses was measured for the indicator 'mire-specific species' for the species level. Numbers of specialised plant formations, special habitats and integration into peatland biotope networks were measured for the indicator 'spatial structure', representing the biocoenosis level. The degree of topsoil degradation and water table depth were measured for the indicator 'site characteristics', representing the ecosystem level. Our assessment via the indicator system was consistent with the assessments of experts (n=11) and practitioners (n=42) in peatland science, nature conservation and forestry.

The few comparable approaches assessing mire biodiversity focus mainly on fauna or vegetation and are often not completed (Tiemeyer et al. 2015; Joosten et al. 2015; Görn & Fischer 2011). Tiemeyer et al. (2015) and Joosten et al. (2015) suggest the assessment of mire biodiversity based on biotope values, and Görn & Fischer (2011) suggest an assessment of fauna, albeit, as fauna specialists are rare, we argue that this is not easily applicable. Further, literature on mire biodiversity (Bragg & Lindsay 2003; Prentice 2011; Minayeva et al. 2017; Strobel et al. 2019) argues that any assessment of mire biodiversity needs to be multifunctional and include heterogeneity on all levels. Through addressing different levels of biodiversity, the indicator system used to assess mire-specific biodiversity not only helps to differentiate between near-natural and degraded peatlands, but also to rate different stages of degradation gradually. Similarly, different stages following restoration are reliably represented, as shown in this thesis. This is supported by Luthardt et al. (2023), who show, by using the indicator system, that although an improvement in biodiversity values can be detected, restored peatlands differ from mires that have matured over time and with high continuity. Tanneberger et al. (n.d.) apply the indicator system in the context of quantifying the ecosystem services of rewetted peatlands and illustrate that it allows scenario building to estimate the development of mire-specific biodiversity after peatland rewetting.

The indicator system can best be applied using the vegetation form concept (Koska 2001), along with soil sampling to determine peat degradation in each vegetation unit and the additional notation of special habitats in the field. An assessment sheet was developed to allow easy application, which can be filled in during fieldwork. The indicator system can be adapted elsewhere by following the approach of Hammerich et al. (2022). First, reference systems (see Wagner & Wagner (2004) or Mendes et al. (2019), as examples of how to cluster peatlands based on naturalness) need to be analysed for their mire-specific components, especially species, plant formations, special habitats and integration into the biotope network. Vascular plants and moss species native to mires, for example, are provided by Joosten et al. (2017a) for European countries. Plant formations (Ellenberg & Müller-Dombois 1966) are a global concept and can be adapted to how they are regionally manifested. As mires worldwide share the same hydrological processes (Parish et al. 2008), good transferability in soil and water table assessments is expected.

GHG emissions

GHG emission factors are generally based on CO_2 flux measurements, measured by chamber technique or eddy covariance method (Poyda et al. 2017). Building on existing data, the GEST approach has been employed to meta-analyse annual GHG fluxes in relation to the peatland site parameters water table, soil type, vegetation composition, trophic condition and acidity, with the end result being that mean annual water table values best explain CO_2 and CH_4 fluxes. Vegetation is used as a proxy to indicate soil moisture classes with mean annual groundwater tables (vegetation forms by Koska (2001)) and to allocate GEST types accordingly (Couwenberg et al. 2011). We used the GEST values provided by Reichelt (2015), who updated the original work done by Couwenberg et al. (2011) for open vegetation units. Spangenberg (2011) was applied for forested vegetation units. New publications on GEST values are being prepared, as the work by Spangenberg (2011) was interpolated from open GESTs and has shown some inaccuracies. Furthermore, newly measured flux data from 2015 onwards will be included to revise GESTs for open vegetation types.

The GEST approach is applicable for lowlands in north-west Europe. For other climatic and biogeographical zones, calibration is needed – as attempted for Belarus (Tanneberger & Wichtmann 2011) and the Baltic states (Jarašius et al. 2022). Calibration seems possible, as studies on GHG emissions from peatlands and measuring flux data have increased over the past years, being the main research topic in peatland science (Van Bellen & Larivière 2020). Generally, regular updates of GEST values are necessary to reflect the current state of knowledge. I argue that measuring fluxes for every restoration project will not be possible, especially over a long period, as flux measurements are expensive and require significant expert knowledge. Moreover, they usually do not represent all areas in a peatland. Therefore, interpolation and meta-analyses will be necessary to estimate GHG emissions. In this context, the GEST approach presents a robust methodology and an alternative to direct flux measurements. The main challenge for transferability is the use of the underlying vegetation typology, which is currently only applicable in north-east Germany (see above).

5.3. Restoration effects

The research question addressed in the following chapter:

III b How and to what extent do water tables, peat-accumulating areas, mire-specific biodiversity and GHG emissions change from before to after restoration?

The restoration projects assessed in the context of this thesis illustrate that peatland restoration generally leads to rises in water table depth, increased TOC values comparable to undisturbed peats and an increased peat accumulation area, an increase in mire-specific biodiversity and an estimated decrease in GHG emissions. The chosen monitoring parameters detected changes reliably in these peatland properties. However, peatlands are mostly not restored to near-natural conditions within the studied timeframe, as confirmed in comparable studies (e.g., Mälson et al. 2008; Krejčová et al. 2021; Kreyling et al. 2021). Pristine mires are characterised by a high degree of maturity and continuity (in the study region, up to 13,000 years) and, as a result, largely stable conditions in terms of species composition, nutrient ratios, vegetation zoning, water tables, structural diversity and peat formation over centuries (cf. Parish et al. 2008; Luthardt 2014a; Wulf 2001). Therefore, mires are mostly resilient to natural disturbances

(Meier-Uhlherr et al. 2014). Once this continuity is interrupted and a certain threshold is reached, turning them back to their former selves is questionable and not achievable in the short term (Loisel & Gallego-Sala 2022). This does not mean that peatlands cannot generally be restored to pristine conditions, but it could take a long time – evidently longer than the relatively short timeframe of peatland restoration and its monitoring. As such, the goal of peatland restoration should be to enhance ecosystem functioning (cf. Chimner et al. 2016). The closer a peatland is to undisturbed conditions, the higher its resilience to disturbances, which is especially relevant in the context of adapting to climate change (Meier-Uhlherr et al. 2014; Loisel & Gallego-Sala 2022; Luthardt et al. 2023). Additionally, the positive effects of enhanced ecosystem functioning of rewetted peatlands for adjacent ecosystems need to be accounted for, such as increased water retention and, therefore, water availability for forests in the peatland catchment areas or the enhancement of water quality and biodiversity of water bodies (cf. Ramchunder et al. 2012), which additionally promote peatland restoration.

Water table

Through restoration, the water table could be raised in the majority of studied peatlands. By using the soil moisture class methodology, it could be distinctly determined that only about 25% of the 'Protection programme for forest peatlands in Brandenburg' study sites showed a water table at the surface in at least 90% of the peatland area. Two of the six Stechlinsee peatlands could be fully rewetted with a surface-level water table in 100% of the peatland area (Luthardt et al. 2021). Although the water table can successfully be raised (e.g., Haapalehto et al. 2011; Laine et al. 2011), it is challenging to fully rewet peatlands due to the high anthropogenic alteration of the majority of peatlands and their catchment areas (Convention on Wetlands 2021). Near-surface peat degradation, and therefore the loss of a functioning, self-regulating acrotelm, altered surface structures, changes from vegetation with high water storage capacity to ubiquitous species and interference with the groundwater balance on a landscape scale are known to influence rewetting success negatively (e.g., Grootjans & van Diggelen 1995; Haapalehto et al. 2011; Hedberg et al. 2012; Strobel et al. 2019; Kreyling et al. 2021). However, especially in peatlands, where near-natural areas remained, successful rewetting by hydraulic measures led to a quick re-colonisation of peat-forming vegetation into the degraded areas, resulting in the new formation of the acrotelm (see also below on peat properties and accumulation). The acrotelm's significant water storage capacity and vertical gradient in terms of hydraulic conductivity stabilise the water table and are seen as key challenges in restoring peatlands to self-regulating mires (Convention on Wetlands 2021).

Peat properties and accumulation

The research questions addressed in the following section:

II a How do TOC, TN, C/N ratio, pH value, BD and structure – as important indicators in the recovery of near-surface peat and carbon sink function – change, from before to after restoration?

II b Are the results comparable to pristine site values (based on a comparison of the literature)?

The peat properties pH-value, TOC, TN, C/N ratio and BD were sampled before and 17 years after restoration in the Stechlinsee peatlands. PH value remained stable over time. The mainly significantly increased values in TOC, from a mean of 35.7% to 47.8% in 2002–2004 (before restoration) to a mean of 42.5% to 54.0% in 2019–2021 (after restoration), are comparable to values of undisturbed peats. After restoring Stechlinsee peatlands, TOC values are 42.5% to 50.4% for sedge-dominated peatlands and 47.7% to 54% in peat moss-dominated sites. For north-east Germany, Succow (1988) states a mean of 40.8% TOC for non-degraded sedge peat and 48.6% TOC for non-degraded moss peats. For peatlands in the northern hemisphere, Loisel et al. (2014) determined for herbaceous peat, including sedge peats, a mean of 50.5% TOC and for Sphagnum peat, 46% TOC.

Furthermore, a decrease in nutrient availability was determined, highlighted by a dominant increase in the C/N ratio. This resulted from a slight decrease in TN values at four sites (one significant) and, at the same time, an increase in TOC. A decrease in nutrient availability was also determined at the 'Protection programme for forest peatlands in Brandenburg' study sites, indicated by the change in vegetation to species with lower nutrient demands. This decrease is expected to positively influence the diversity of mire-specific plant species, which are often adapted to moderately to very nutrient-poor conditions (cf. Minayeva et al. 2017).

A mainly significant decrease in BD was detected, ranging from 0.08 g/cm^3 to 0.48 g/cm^3 in 2002–2004 and from 0.10 g/cm^3 to 0.16 g/cm^3 in 2019–2021. Values after restoring sedge-dominated sites are comparable to those for undisturbed herbaceous peat with a mean of 0.12 g/cm^3 (Loisel et al. 2014). The BD of peat moss-dominated sites was twice as high as the mean of 0.08 g/cm^3 given by Loisel et al. (2014) for *Sphagnum* peat in the northern hemisphere. As there is a clear relation between high C content and low BD, the decrease in BD is likely due to a rise in C (Chapman et al. 2017; Wittnebel et al. 2021). The loosening of peat by buoyancy most likely led to a further decrease in BD at the site, which had the highest decrease in this factor.

A structural change was visible in the degraded, rewetted near-surface peats. The structure of the earthified and murshified near-surface peats changed to a more homogenous sludge mass showing larger re-aggregates. Additionally, in peat moss-dominated areas, a layer of low decomposed peat moss peat accumulated on top of the older, degraded peat horizons. This new layer was clearly distinguishable by colour and structure compared to the degraded layer beneath. The new peat moss layers at three Stechlinsee sites had a mean thickness of 6.8 cm to 36.1 cm and a mean accumulation rate of 0.38 to 2.01 cm/y. At the 'Protection programme for forest peatlands in Brandenburg' study sites, new peat moss layers had an accumulation rate of up to 1.5 cm/y. In peatlands dominated by herbaceous plants, on the other hand, new material is added at varying depths into an older matrix by vertically ingrown rhizomes and roots (Michaelis et al. 2020). This so-called 'displacement peat' was visible in the sedge-dominated peatlands, where the degraded peat layer functions as this matrix. Both phenomena were determined in both the Stechlinsee peatlands and the 'Protection programme for forest peatlands in Brandenburg' study sites. In addition to the formation of displacement peat, Mrotzek et al. (2020) and Michaelis et al. (2020) describe a 'proto peat' 20 years following restoration, consisting of newly deposited roots, radicals and litter on top of the degraded peat in sedge-dominated areas. In very few, very wet sedge-dominated areas in the 'Protection programme for forest peatlands in Brandenburg' study sites, we determined a very thin layer of newly accumulated litter, mainly consisting of above-ground plant parts of tall sedges, which did not appear to be peat at that point but has the potential to become peat when continuous wet conditions prevail. These developments led to the conclusion that within 20 years of restoration, the formation of a new acrotelm is possible.

However, the results shown herein need to be interpreted with respect to ongoing decomposition, especially in the highly dynamic near-surface peat layer. The added material and TOC might never, or only partially, become part of the long-term C store (Young et al. 2019). This is an especially important notion in the context of climate change, in terms of potentially longer dry periods in the summer and expected longer periods of aerobic conditions in the topsoil peat (cf. DWD 2019).

Mire-specific biodiversity

By using the indicator system to assess mire-specific biodiversity, we determined a general increase in its value after restoration. This increase mainly resulted from higher water tables, increased peat-accumulating area, and habitat heterogeneity. Mire-specific species, on the other hand, increased just slightly and not significantly after restoration. The parameters and measure values of the indicator system well represented these different developments in the different biodiversity levels following restoration. High mire-specific biodiversity values are rarely achieved, as supported by Tanneberger et al. (n.d.), who established – also by using the indicator system – that an increase in biodiversity following restoration did not reach high or even projected moderate values. In general, the biodiversity of mires, which is commonly assessed in relation to vegetation or species alone, recovers slowly and is not comparable to near-natural conditions (Benayas et al. 2009; Hedberg et al. 2012; Kareksela et al. 2015; Krejčová et al. 2021; Mälson et al. 2008). For fen systems in temperate Europe, helophytisation, i.e., a shift towards tall-growing, species-poor vegetation, was observed (Kreyling et al. 2021).

Only when biodiversity value was still high before restoration, representing a low degree of degradation or near-natural, often central areas remained within the peatland, functioning as a starting point for the re-colonisation of mire-typical and mire-specific vegetation and the formation of a new acrotelm, were high biodiversity values reached. Similar findings on the restoration of fen systems support these results, as the potential for restoring them to near-natural conditions is higher, the lower the degree of initial degradation and change in vegetation (Grootjans & van Diggelen 1995; Klimkowska et al. 2010; Laine et al. 2011). The shift towards lower nutrient availability might also positively influence the re-colonisation of mire-specific species in the long term, which are often adapted to extreme nutrient conditions (cf. Minayeva et al. 2017). In this context, additional long-term monitoring data is needed.

GHG emissions

For the 'Protection programme for forest peatlands in Brandenburg' study sites, we estimated GHG emissions via the GEST approach (Couwenberg et al. 2011; Spangenberg 2011; Reichelt 2015). In the reduced sample of 29 study sites, where peatlands with a negative influence of groundwater abstraction in the peatland catchment areas were excluded, GHG emissions decreased from before to after restoration from a median of 8.5 to 6.8 t CO_2-eq. ha^{-1} $year^{1}$. This reduction was higher in peatlands undergoing hydraulic measures, from a median of 10.7 to 7.1 t CO_2-eq. ha^{-1} $year^{-1}$. It was lower and not significant in peatlands undergoing management measures, from a median of 7.5 to 6.6 t CO_2-eq. ha^{-1} $year^{-1}$. The estimated GHG reduction at 80% of the study sites ranged between 0.15 to 16.19 t CO_2-eq. ha^{-1} $year^{-1}$. The slight increases in GHG emissions of 0.13 to 3.87 t CO_2-eq. ha^{-1} $year^{-1}$ in the remaining 20% were coherent with a slightly deeper water table or no apparent water table rise (n=3) and/or a change in vegetation with a shift towards shunt species. All study sites contributed GHG emissions after restoration, with values ranging from 2.32 to 16.27 t CO_2-eq. ha^{-1} $year^{-1}$.

Our estimated values for GHG reduction are comparable, but slightly lower, than the values provided by Wilson et al. (2016) for GHG emission reduction through peatland rewetting in the temperate zone: for grassland 4.68 t CO_2-eq. ha^{-1} $year^{-1}$ (nutrient-rich, shallow drained) to 20.84 t CO_2-eq. ha^{-1} $year^{-1}$ (nutrient-poor), and for forest land -0.32 t CO_2-eq. ha^{-1} $year^{-1}$ (nutrient-rich) to 8.47 t CO_2-eq. ha^{-1} $year^{-1}$ (forest land, nutrient-poor). This can be related to the less intensive usage – and therefore less intensive drainage – of the smaller, hardly accessible forest peatlands in the study area compared to larger areas used for agriculture (cf. Kühn et al. 2014).

In general, the restoration of the 'Protection programme for forest peatlands in Brandenburg' study sites led to significant reductions in GHG emissions compared to their drained states, but all peatlands remained contributors of GHG emissions, which is comparable to the literature (Couwenberg et al. 2011; Wilson et al. 2016; Humpenöder et al. 2020). In this context, dissimilar vegetation composition after restoration, e.g., the absence of brown mosses and a shift to graminoid wetland plants in fens, is suspected to influence the total GHG effects of rewetting (Zak et al. 2015; Kreyling et al. 2021). For the Stechlinsee peatlands, although no GHG emissions were estimated, we suspect a reduction in GHG emissions due to higher water table depths at the majority of the study sites.

However, in this respect, it is important to state that the GEST approach can result in some uncertainties (see Chapter 5.3). Furthermore, potentially high methane emissions (cf. Antonijević et al. 2023), directly after hydraulic measures were applied, were not accounted for, as the study sites were mapped before and 4 to 27 years after restoration. Whereas some studies revealed high methane emissions following rewetting, potentially lasting decades (Zak et al. 2015; Antonijević et al. 2023), others argue that these methane emissions do not counteract the climate change mitigation potential of peatland rewetting in the long term (Couwenberg et al. 2011; Günther et al. 2020). In this case, then, further long-term monitoring data is required.

5.4. Factors influencing restoration success

Research question addressed in the following chapter:

III c How do restoration measures, groundwater-abstracting facilities in the peatland catchment areas, trophic conditions, the acid-base ratio before restoration, hydrogenetic mire type, peatland size and years since restoration influence restoration success?

We demonstrated that variables indicating the success of peatland restoration, i.e., higher water tables, an increase in the peat-accumulating area and mire-specific biodiversity as well as lower GHG emissions, are highly related. Whereas variables such as trophic conditions and the acid-base ratio before restoration, hydrogenetic mire type, peatland size and changes in trophic conditions are not related to restoration success, we found a positive relation with years since restoration. The few comparable studies support the contention that time since restoration can positively influence mire-typical vegetation and plant diversity (Karksela et al. 2015; Strobel et al. 2019). Comparable studies on the influence of trophic conditions and the acid-base ratio, hydrogenetic mire type and peatland size are rare, in which case it is indicated that peatlands with a percolating hydrological regime are especially hard to restore, due to specific water table dynamics. Alkaline fens are among the most threatened mire types, and successful restoration requires intensive measures, such as topsoil removal and diaspor transfer (Klimkowska et al. 2007; Nilson 2015). In respect to peatland size, Heikkilä & Lindholm (1995) argue that small peatlands, aligned with high dependency on their surroundings, are generally harder to restore, especially in regions with a negative climatic water balance. Both, Stechlinsee peatlands and 'Protection programme for forest peatlands in Brandenburg' study sites are highly dependent on the surrounding catchment area, as the climatic water balance in Brandenburg is negative (cf. Luthardt & Zeitz 2014) and they are small, mostly under 6 ha in size. In respect to the 'Protection programme for forest peatlands in Brandenburg' study sites, groundwater abstracting facilities significantly hindered restoration success, as supported, for example, by Grootjans & van Diggelen

1995, who have stated that successful fen restoration can only be achieved on a landscape scale.

We found great differences in restoration outcomes rooted in the applied restoration measures. Management measures, for instance, did not significantly improve peatland conditions but rather maintained the initial state. In contrast, hydraulic measures led to a rise in the water table, mire-specific biodiversity and peat-accumulating area as well as lower estimated GHG emissions. This contrasting restoration outcome is based on the difference in peatland conditions prior to initiating restoration measures. In this regard, peatlands undergoing management measures were in a better initial state (mire-specific biodiversity of 11 points), showed no active drainage ditches and management measures were the only option for action. Peatlands undergoing hydraulic measures were in a worse initial state (mire-specific biodiversity of 9 points) and actively drained by ditches, thereby showing a greater potential for improvement. A higher water table especially is attained in the short term, which then kickstarts the re-colonisation of peat-accumulating species, peat accumulation and a more diverse habitat. The increase in mire-specific plant species seems to be long term. Hedberg et al. (2012) revealed that rewetting, in combination with clear cutting, is effective in restoring wetland vegetation and mowing is estimated to enhance successful fen vegetation recovery (Grootjans & Van Diggelen 1995; Mälson et al. 2008) which indicates that management, in combination with prior hydraulic measures, has highest restoration success prospects. Generally, it is possible that the conceivable positive effects of management measures have been counteracted by climate change and reduced water supply in recent decades in the study area. Furthermore, forest restructuring in the peatland catchment areas is a measure via which only long-term effects can be expected. The effects of different management measures on restoration success require further multi-case research encompassing long-term data.

6. Conclusion with implications for practice

- As it is challenging to restore peatlands to near-natural conditions, the first priority needs to be the conservation and stabilisation of all pristine mires for their high biodiversity value and several ecosystem functions. Second, as peatland rewetting generally leads to a rise in the water table, increases in the peat-accumulating area, increases in mire-specific biodiversity and a decrease in GHG emissions, it is necessary to rewet all degraded peatland as one key factor in mitigating climate change and fighting biodiversity loss. As GHG emissions resulting from human activities have already had negative effects on the diverse ecosystems of the Earth, and global warming is more likely than not to reach 1.5 °C in the near term (IPCC 2023), peatland restoration is more urgent than ever in getting closer to meeting the goals of the UNFCCC and the Paris Agreement. International efforts to stop biodiversity loss have fallen short of meeting set targets (XU et al. 2021), and the conservation of mires and restoration of peatlands is necessary to achieve the 2050 vision of the post-2020 Global Biodiversity Framework of CBD.
- We should ask if the goal of peatland restoration should be to reconstruct past conditions or rather to assist peatland ecosystems in increasing ecological functioning (cf. Chimner et al. 2016). Even if near-natural conditions are not always attained by restoration, it is evident that rewetting by hydraulic measures enhances ecosystem functioning and favours wetland species (see also Luthardt et al. 2021; Tanneberger et al. n.d.). Especially in the context of adaptation to climate change, the priority must be to strengthen peatland ecosystems' resilience by moving them closer to undisturbed conditions. Moreover, the positive effects for adjacent ecosystems, such as increased water retention and water availability for forests in peatland catchment areas, highlight the importance of peatland restoration.
- As rewetting fully is a challenging undertaking, but is nonetheless the first step in successfully restoring peatlands, the focus of restoration efforts needs to be on how to achieve this outcome. In this context, water availability on a landscape scale, aligned with land-use in catchment areas, is of major importance. In respect to forest peatlands, this means, among others, restructuring forests back to near-natural conditions.

- Although management measures have not led to significant improvements in the context of our study sites, they should be considered for maintaining peatland conditions or even enhancing peatland restoration success in addition to hydraulic measures (cf. Hedberg et al. 2012). Forest restructuring within peatland catchment areas, as a long-term measure, should be implemented not only for restoration areas but also for largely functioning mires to enhance water availability in the context of future shortages in water availability due to global warming. As forest restructuring has a medium- to long-term effect, it must be realised as quickly as possible. Therefore, it is also advisable to start with forest restructuring if restrictions on hydraulic measures or groundwater-reducing forms of land use in the catchment area exist. If the restrictions or negative effects are lifted, the positive effects of forest restructuring can already be effective.
- After successful rewetting and, therefore, a water table close to the surface, we determined an increase in TOC in near-surface peats comparable to values of undisturbed peats. Therefore, the re-establishment of peatlands as C sinks seems possible, as TOC in near-surface peats is a key factor in this process. Nevertheless, due to ongoing decay in highly dynamic near-surface peat, and potential negative effects due to climatic change, transfer into the long-term C store is questionable. Still, the decrease in BD and newly accumulated peat-moss layers indicate the development of a new acrotelm, which is a key factor in restoring peatlands to self-regulating mires, due to their capability to stabilise the water table.
- Whereas changes in the water table can be achieved in the short term, leading to the expansion of peat-forming vegetation and, therefore, new peat accumulation and increased habitat heterogeneity, the recovery of mire-specific species is rather slow. The mid-term recovery of mire-specific biodiversity can be expected from comparably low-degraded peatlands when near-natural areas within the peatland remain. Additionally, decreased nutrient availability is expected to support the re-colonisation of mire-specific species in the long term.
- Although restored peatlands remain contributors of GHG emissions, a reduction compared to their drained state was found, which highlights the importance of peatland restoration for climate change mitigation. That said, uncertainties still

remain in relation to the formation of shallow water bodies and, therefore, rather long-lasting methane emissions following rewetting, which counteract GHG emission savings through preserving the peat body.

- Water tables, peat accumulation, mire-specific biodiversity and GHG emissions, assessed with the proposed or comparable methodologies suggested herein, should be considered as standard monitoring parameters for controlling the success of peatland restoration projects in the study region and other geographical regions. As most assessments are based on spatial vegetation mapping and topsoil sampling, our proposed monitoring tools are relatively cost-effective and produce robust results. The collected information can be used to adapt restoration practices and enhance restoration success. Science-based and robust data is needed to justify the use of public funds and in the context of raising efforts to quantify ecosystem services (cf. La Notte et al. 2017; SBTN 2023). Additionally, if restoration aims require and project resources allow, additional sampling is recommended, e.g., peat properties (TOC, TN, C/N ratio, BD, pH value). A holistic overview in this regard is provided by Reed et al. (2022) and can be used as a 'tool box' for choosing monitoring parameters.
- The novel indicator system employed to assess mire-specific biodiversity, based on the specific characteristics of mires, highlights the importance of peatland areas for biodiversity in general and provides information for further planning and adapting conservation practices. In the context of payments for ecosystem services relating to peatland rewetting, as shown in Tanneberger et al. (n.d.), the indicator system can quantify mire-specific biodiversity.

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III Acknowledgements

I feel fortunate to be a peatland scientist. Over the past years, I have seen beautiful areas in Brandenburg, wading through moss carpets, sometimes up to my belly in water, finding rare and endangered species, and taking refreshing swims in clear lakes after a long day of work. Apart from the regional peatland beauty, I have experienced many conservation practices in national and international networks in Germany, the Netherlands, South Africa, and Russia. It is rewarding to work in a field where positive progress towards a more sustainable future is visible and to work with people who share a passion for peatlands and nature. Now, I am happy and grateful to acknowledge the people who supported me during the years of my PhD.

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VI APPENDIX

1. List of publications by Jenny Hammerich
2. Overview of articles of this dissertation with author contributions
3. Abstracts of co-authored peer-reviewd publications during the time of PhD
4. Declarations and Assurances (*Erklärungen und Versicherung*en)

Appendix 1: List of publications by Jenny Hammerich

Peer-reviewed

Hammerich J, Schulz C, von Wehrden H, Zeitz J, Luthardt V (n.d.) Monitoring peatland restoration in forests – The effects of hydraulic and management measures on the water table, peat accumulation, mire-specific biodiversity and greenhouse gas emissions. *Restoration Ecology (submitted April 2024)*

Dabard CH, Gohr C, Weiss F, von Wehrden H, Neumann F, Hordasevych S, Arieta B, **Hammerich J**, Meier C, Jargow J, Luthardt V, Ibisch PL, Ferreira AF (n.d.) Biosphere Reserves as model regions for transdisciplinarity? A literature review. *Sustainability Science (in review, submitted December 2023)*

Tanneberger F, Berghöfer A, Brust K, **Hammerich J**, Holsten B, Joosten H, Michaelis D, Moritz F, Reichelt F, Schäfer A, Scheid A, Trepel M, Wahren A, Couwenberg J (n.d.) Quantifying ecosystem services of rewetted peatlands – the MoorFutures methodologies. *Ecological Indicators (in review, submitted October 2023)*

Hammerich J, Schulz C, Probst R, Lüdicke T, Luthardt V (2024) Carbon content and other soil properties of near-surface peats before and after peatland restoration. *PeerJ (in press)*

Luthardt V, Brauner O, **Hammerich J**, Probst R, Schulz C, Wachtel S, Luthardt V (2023) Resilienz naturnaher Moore im Klimawandel – Fallbeispiele aus dem Biosphärenreservat Schorfheide-Chorin (Resilience of near-natural peatlands in times of climate change – case studies from the Schorfheide-Chorin biosphere reserve). *Natur und Landschaft* 98/3:124–131. In German. DOI: 10.19217/NuL2023-03-04

Hammerich J, Dammann C, Schulz C, Tanneberger F, Zeitz J, Luthardt V (2022) Assessing mire-specific biodiversity with an indicator based approach. *Mires and Peat* 28/32:1–29. DOI: 10.19189/MaP.2021.SJ.StA.2205

Luthardt V, Lüdicke T, **Hammerich J**, Schulz C (2021) Erfolgreiche Revitalisierung naturnaher Moore im Naturpark Stechlin-Ruppiner Land (Successful revitalisation of near-natural peatlands in the Nature Park Stechlin-Ruppiner Land). Pages 169–191 In: Scherfose V (ed) Erfolgskontrollen im Naturschutz (Success Control in Nature Conservation). BfN-Skript 171, Bonn - Bad Godesberg. In German.

Presentations

Hammerich J, Koch F-G (2024) Moorschutz und –renaturierung im Wald (Peatland conservation and restoration in forests). Seminar ‘Renaturierung von Mooren‘. Lions-Club Deutschland. Online. 28 March 2024.

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Hammerich J, Luthardt V (2022) Moorrenaturierung im Wald (Peatland restoration in forests). Waldbaukolloquium. Landesforst Brandenburg. Eberswalde (Germany). 27.09.2022.

Hammerich J, Luthardt V, Zeitz J (2022) Erfolgskontrolle von Waldmoor-Renaturierungsmaßnahmen – methodisches Konzept und Erfolge der Moorschutzmaßnahmen der Brandenburger Landesforsten (Assessing the success of forest peatland restoration measures – methodological concept and success of peatland restoration of the State Forestry Brandenburg). Bundesfachtagung ‘Wiedervernässung von Waldmooren‘. Sachsenforst. Erzgebirge (Germany). 01 & 02 September 2022.

Luthardt V, Herold B, **Hammerich J** (2022) Peatlands in the central Schorfheide-Chorin biosphere reserve and Sernitz spring mire restoration. Biosphere Reserves Institute Research Conference. Biosphere Reserves Institut. Altenhof (Germany). 18 May 2022

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Hammerich J (2021) Moorrenaturierung in Brandenburger Wäldern: Auswirkungen auf die moorspezifische Biodiversität (Peatland restoration in Brandenburg's forests: effects on mire-specific biodiversity). BfN-Veranstaltungsreihe ‘Biodiversität und Klima – Vernetzung der Akteure in Deutschland‘. Internationale Naturschutzakademie Vilm. Vilm (Germany). 08 September 2021.

Hammerich J, Luthardt V (2020) Wirkung von bis zu 20 Jahren Moorvernässung in den Brandenburger Wäldern (Effect of up to 20 years of peatland rewetting in Brandenburg's forests). DGMT-Jahrestagung. Deutschen Gesellschaft für Moor & Torfkunde. Rendsburg (Germany). 27 September 2020.

Hammerich J (2019) Moore als Oasen einheimischer Vielfalt – Einblicke und Bewertungsansatz (Peatlands as oases of native diversity – insights and assessment approach). Öffentliche Vortragsreihe ‘Umwelt und Nachhaltigkeit‘. Universität Hildesheim. Hildesheim (Germany). 17 April 2019.

Hammerich J, Luthardt V, Zeitz J (2018) Projekt Erfolgskontrolle Waldmoorrentaturierung – Moorschutz im Landeswald Brandenburg bald in einer neuen Qualität (The project ‘Monitoring forest peatland restoration‘ – a new quality of peatland conservation in the state forests of Brandenburg soon to be realised). 4. Waldnaturschutztag. Landesbetrieb Forst Brandenburg. Eberswalde (Germany). 19 November 2018.

Hammerich J, Luthardt V (2018) Die Biodiversität Brandenburger Moore – methodische Annährung und Fallbeispiele (The biodiversity of peatlands in Brandenburg - methodological approach and case studies). Tagung ‘Moore in Europa – Bestand, Bedrohung, Schutz und Wiederherstellung’. Brandenburgische Akademie Schloss Criewen. Criewen (Germany). 18 & 19 October 2018.

Luthardt V, Schulz C, **Hammerich J**, Dammann C (2017) Catalog of the typical and specific plant species of mires including mosses of Brandenburg and Berlin and its application for the evaluation of the biodiversity of mires. Flora & Funga Pomeranica – The Third Symposium on Flora, Fungy, Vegetation and Landscape of Pomerania. Szczecin (Poland). 13–15 September 2017.

Hammerich J, Dammann C (2017) How to measure the biodiversity of peatlands – an indicator-based approach. LIFE Peat Restore Workshops ‘Peatlands and Atmosphere’. NABU-Berlin. Klein Köris (Germany). 27–30 April 2017.

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Hammerich J, Luthardt V, Zeitz J (2022) Moorschutz in den Wäldern Brandenburgs – Waldmoorschutzprogramm, Erfolge, Renaturierungsmaßnahmen (Peatland Protection in the Forests of Brandenburg – Forest Peatland Protection Programme, Successes, Restoration Measures. Hochschule für Nachhaltige Entwicklung Eberswalde, Eberswalde. In German. DOI: 10.57741/opus4-366

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Luthardt V, Brauner O, **Hammerich J**, Probst R, Schulz C, Wachtel SF (2022) Steckbriefe und weiterführende Informationen zu ausgewählten Mooren des Biosphärenreservats Schorfheide-Chorin (Brief Profiles and Further Information on Selected Peatlands in the Schorfheide-Chorin Biosphere Reserve). Hochschule für Nachhaltige Entwicklung Eberswalde, Eberswalde. In German. DOI: 10.57741/opus4-377

Gather C, **Hammerich J** (2017) Moore in landwirtschaftlicher Nutzung: Hotspots des Klimwandels (Peatlands in agricultural use: hotspots of climate change). Pages 14–20 In: Umweltbundestamt (eds) Umweltschutz in der Landwirtschaft (Environmental Protection in Agriculture). Umweltbundesamt, Dessau-Rosslau. In German.

Appendix 2: Overview of articles of this dissertation with author contributions

Study I:	**Assessing mire-specific biodiversity with an indicator based approach**
Authors:	J. Hammerich, C. Dammann, C. Schulz, F. Tanneberger, J. Zeitz, V. Luthardt
Journal:	Mires and peat (Two-year impact factor: 1.488)
Authorship:	First author with predominant contribution
Author contributions:	JH, CD, VL and FT designed the study. JH and CD did the literature review and set up the first draft of the indicator system. JH revised the indicator system and sampled the data. The data were collected by JH, CD, CS and other researchers at Eberswalde University for Sustainable Development. JH prepared the manuscript with the help of CS and VL. VL critically reviewed the study and contributed central ideas and discussion points. All authors contributed to the final version of the manuscript.
Publication status:	Published (October 2022)
Study II	**Carbon content and other soil properties of near-surface peats before and after peatland restoration**
Authors:	J. Hammerich, C. Schulz, R. Probst, T. Lüdicke, V. Luthardt
Journal:	PeerJ (Two-year impact factor: 3.061)
Authorship:	First author with predominant contribution
Author contributions:	Jenny Hammerich conceived and designed the experiments, performed the experiments, analyzed the data, prepared figures and/or tables, authored or reviewed drafts of the article, and approved the final draft; Corinna Schulz conceived and designed the experiments, performed the experiments, analyzed the data, prepared figures and/or tables, authored or reviewed drafts of the article, and approved the final draft; Robert Probst conceived and designed the experiments, performed the experiments, analyzed the data, prepared figures and/or tables, and approved the final draft; Thomas Lüdicke performed the experiments, prepared figures and/or tables, and approved the final draft; Vera Luthardt conceived and designed the experiments, authored or reviewed drafts of the article, and approved the final draft.
Publication status:	In press (submitted May 2023)
Study III:	**Monitoring peatland restoration in forests – The effects of hydraulic and management measures on the water table, peat accumulation, mire-specific biodiversity and greenhouse gas emissions**
Authors:	J. Hammerich, C. Schulz, H. von Wehrden, J. Zeitz, V. Luthardt
Journal:	Restoration Ecology (Two-year impact factor: 4.181)
Authorship:	First author with predominant contribution
Author contributions:	JH, JZ and VL designed the study. JH screened and reprocessed the data before restoration. JH collected field data after restoration from 2018 to 2020. JH and HvW compiled the statistical analyses. JH wrote the manuscript. CS, HvW, JZ and VL reviewed the manuscript. All authors contributed to the final version.
Publication status	Submitted (April 2024)

Appendix 3: Abstracts of co-authored peer-reviewed publications during the time of PhD

Biosphere Reserves as model regions for transdisciplinarity? A literature review.

Sustainability Science (in review, submitted December 2023)

Caroline Hélène Dabard [1,2†*], Charlotte Gohr [1,3,4†], Fabio Weiss [1,3,5], Henrik von Wehrden [6], Frederike Neumann [1], Solomiia Hordasevych [1], Bruno Arieta [1], **Jenny Hammerich** [1,3], Caroline Meier [1,8], Janine Jargow [1,7], Vera Luthardt [1], Pierre L. Ibisch [1,4], Ana Filipa Ferreira [1]

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Abstract

The UNESCO Man and the Biosphere program advocates and designates Biosphere Reserves as learning sites for sustainable development. Yet the extent to which research aligns with their core objectives - biodiversity conservation, economic development and capacity building - remains uncertain. In response, transdisciplinary research in conservation and development aims at implementing more diverse, participatory methods to improve effective management as well as governance. This study provides a systematic screening of scientific research in and on Biosphere Reserves published since 1975. Research fields in Biosphere Reserves are diverse and range from social to political to ecological investigations. We identified an emerging field of transdisciplinary science in research related to or conducted in UNESCO Biosphere Reserves, highlighting progress in author gender equality as compared to studies that did not build on a transdisciplinary mode. Most transdisciplinary studies were conducted in Mexican and Indian Biosphere Reserves. Transdisciplinary research in Biosphere Reserves calls for high-impact knowledge, addressing deep leverage points and the inclusive participation of underrepresented and discriminated groups. Thereby, Biosphere Reserves as specialized areas for sustainable development could play a vital role.

Key words

Systematic literature review, leverage points, knowledge types, transformative research, participation

Quantifying ecosystem services of rewetted peatlands – the MoorFutures methodologies

Ecological Indicators (in review, submitted October 2023)

Franziska Tanneberger[1,2*], Augustin Berghöfer[2], Kristina Brust[3], **Jenny Hammerich**[4,5], Bettina Holsten[7], Hans Joosten[1], Dierk Michaelis[1], Fiedje Moritz[1], Felix Reichelt[6], Achim Schäfer[1,6], Aaron Scheid[8], Michael Trepel[7], Andreas Wahren[3], John Couwenberg[1,6]

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Abstract

In 2011, MoorFutures® were introduced as the first standard for generating credits from peatland rewetting. We developed methodologies to quantify ecosystem services before and after rewetting with a focus on greenhouse gas emissions, water quality, evaporative cooling and mire-typical biodiversity. Both standard and premium approaches to assess these services were developed, and tested in the rewetted polder Kieve (NE-Germany). The standard approaches are default tier 1 estimation procedures, which require little time and few, mainly vegetation data. Based on the Greenhouse gas Emission Site Type

(GEST) approach, emissions decreased from 1,306 t CO_2e in the baseline scenario to 532 t CO_2e in the project scenario, whereas 5 years after rewetting they were assessed to be 543 t CO_2e per year. Nitrate release assessed via Nitrogen Emission Site Types (NEST) was estimated to decrease from 1,088 kg N (baseline) to 359 kg N (project), and appeared to be 309 kg N per year 5 years after rewetting. The heat flux - determined with Evapotranspiration Energy Site Types (EEST) – decreased from 6,691 kW (baseline) to 1,926 kW (project), and was 2,250 kW per year 5 years after rewetting. Mire-specific biodiversity was estimated to increase from very low (baseline) to high (project), but was only low 5 years after rewetting. The premium approaches allow quantifying a particular ecosystem service with higher accuracy by measuring or modelling. The approaches presented here have been elaborated for North-Germany but can be adapted for other regions. We encourage scientists to use our research as a model for assessing peatland ecosystem services and biodiversity in other geographical regions. Using vegetation mapping and indicator values derived from meta-analyses is a cost-efficient and robust approach to inform payment for ecosystem services schemes and to support conservation planning at regional to global scales.

Key words

Organic soil, peatland restoration, greenhouse gas emission, proxy, bioindication

Resilienz naturnaher Moore im Klimawandel – Fallbeispiele aus dem Biosphärenreservat Schorfheide-Chorin

(English title: Resilience of near-natural peatlands under climate change – Case studies from the Schorfheide-Chorin Biosphere Reserve)

Natur und Landschaft 98/3:124–131. In German. DOI: 10.19217/NuL2023-03-04

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Abstract

Climate change is affecting the few remaining mires that are still accumulating peat. The question thus arising is this: To what extent can the resilience of these autochthonous ecosystems, in all their diversity, be enhanced? For this purpose, long-term observation series of mostly undisturbed peatlands in the Schorfheide-Chorin Biosphere Reserve in the German regional state of Brandenburg are evaluated. These are set in context with the findings of success monitoring of rewetted forest peatlands. A newly developed indicator system for assessing mire-specific biodiversity is used to evaluate the state of the peatland. In addition, greenhouse gas emissions are estimated using the GEST (Greenhouse gas Emissions Site Types) method and potential new peat formation is considered. The analyses show that the buffering capacity of peat accumulating peatlands in the study area is still intact and that disturbances can be overcome without changing the system. The waterlogging measures were consistently successful and led to a measurable revitalisation. The article underscores the urgent need to stabilise the water balance of all peatlands that are still in a near-natural state. This is vital in order to preserve them as important elements of autochthonous biodiversity with all their positive landscape functions.

Key words

Climate change adaptation, climate change impact, dragonflies, monitoring, mire-specific biodiversity, mire-specific vegetation, peatland state

Erfolgreiche Revitalisierung naturnaher Moore im Naturpark Stechlin-Ruppiner Land

(English title: Successful revitalisation of near-natural peatlands in the Nature Park Stechlin-Ruppiner Land)

Pages 169–191 In: Scherfose V (ed) Erfolgskontrollen im Naturschutz (Success Control in Nature Conservation). BfN-Skript 171, BfN, Bonn - Bad Godesberg. In German.

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Abstract

This article summarizes the results of the success control concerning the restoration project of seven mires in the Natural Reserve “Stechlin-Ruppiner Land” within the framework of the EU-LIFE project “Protection and rehabilitation of clear water lakes, mires and swamp forests in the Stechlinsee area”. Based on the initial state, specific development goals were defined and a monitoring program, developed by HNE Eberswalde, was implemented. The methodology is explained in the article. Analysis of the initial state prior to the restoration from 2002 and 2004 allow an assessment of the peatlands type, genesis, historical influence and water conditions. The further development after implementation of the measures was comprehensively documented in the following years (2008, 2013, 2019) based on changes in water level and vegetation characteristics. The condition of the mires 15 years after implementation of the measures is described and the success of the revitalization is evaluated using general mire

characteristics. Finally, recommendations for future methodological approaches in success control are given.

Appendix 4: Declarations and Assurances (*Erklärungen und Versicherungen*)

Hiermit erkläre ich, dass ich mich noch keiner Doktorprüfung unterzogen oder mich um Zulassung zu einer solchen beworben habe.

Ich versichere, dass die Dissertation mit dem Titel „Assessing the effects of peatland restoration" noch keiner Fachvertreterin bzw. Fachvertreter vorgelegen hat, ich die Dissertation nur in diesem und keinem anderen Promotionsverfahren eingereicht habe und, dass diesem Promotionsverfahren keine endgültig gescheiterten Promotionsverfahren vorausgegangen sind.

Ich versichere, dass ich die eingereichte Dissertation „Assessing the effects of peatland restoration" selbstständig und ohne unerlaubte Hilfsmittel verfasst habe. Anderer als der von mir angegebenen Hilfsmittel und Schriften habe ich mich nicht bedient. Alle wörtlich oder sinngemäß anderen Schriften entnommenen Stellen habe ich kenntlich gemacht.

Berlin, 02.04.2024

Jenny Hammerich