Aus der Professur für Bodenphsik

der Agrar- und Umweltwissenschaftlichen Fakultät

Environmental controls on hydrological processes in fen peatlands

Kumulative Dissertation

zur

Erlangung des akademischen Grades

Doktor der Agrarwissenschaften (doctor agriculturae)

an der Agrar- und Umweltwissenschaftlichen Fakultät der Universität Rostock

vorgelegt von

Sate Ahmad, geb. Am 28.07.1987 in Dhaka, Bangladesch
MSc. (Applied Ecology), Master (Resource & Environmental Management)
BSc. (Environmental Science)

Wohnaft in Rostock

Rostock, den ______________

https://doi.org/10.18453/rosdok_id00003354

Promotion

Environmental controls on hydrological processes in fen peatlands

Grunddaten | Betreuer/Gutachter | Einrichtung

Grunddaten

Promovend	Sate Ahmad
Akademischer Grad	Dr. agr.
Promotionsgebiet	Bodenphysik
Verleihungsdatum	12.05.2021
Datum der mündlichen Prüfungsleistung	07.05.2021

Betreuer/Gutachter

Prof. Dr. Bernd Lennartz, Universität Rostock	Betreuer und Gutachter
Prof. Dr. Joachum Schrautzer, Christian-Albrechts-Universität zu Kiel	Gutachter
Prof. Dr. Fereidoun Rezanezhad, University of Waterloo / Canada	Gutachter

Einrichtung

Agrar- und Umweltwissenschaftliche Fakultät (AUF)

Table of Contents

List of Tables

List of Figures

List of Supplementary Figures

Dedication

I dedicate this dissertation:

To my mom: who has always been patient with the number of disciplinary shifts I had to make to find my passion for environmental science; without her support, I would not be where I am today.

To my dear friend and academic guide Mofiz Bhai: without whom I would not have applied to Universities abroad and who made me believe in myself.

To my dear Rostocker friends and girlfriend: who made Rostock feel like home; namely to Weronika for supporting me whenever I have been down with Ph.D. stress, to Rodrigo for always taking /faking interest in my long monologues about Ph.D. research; to Ujmir, who always made me feel that completing a Ph.D. especially with all the silly things I've done in Rostock is a feat like no other; to Andres, who was always there when I needed him; to Sofia my little Italian friend for always being there for me.

To my late botanist Grandmother: one of the strongest persons I have ever known, a true inspiration, and the first female scientist to join Bangladesh Agricultural Research Institute (BARI)

" অজানাকে জানার স্পৃহা মানুষের চিরন্তন। "

"Human's desire to know the unknown is eternal"

- *The Quest for Truth (1959) by Aroj Ali Matubbar, Bangladeshi Philosopher*

Acknowledgements

I would like to acknowledge the support of my Ph.D. supervisor Prof. Dr. Bernd Lennartz for his guidance and supervision throughout the course of this doctoral work. Furthermore, I would like to thank Dr. Haojie Liu, my co-supervisor, who has been pivotal for keeping my sanity, who has given me first-hand knowledge on soil physics, who has always believed in my abilities and is himself a stockpile of peat knowledge. I am grateful to all the collaborators namely Dr. Gerald Jurasinski, Dr. Anke Günther, Florian Beyer, John Couwenberg, Shajratul Alam, and Prof. Dr. Bjørn Kløve who has been key to my progress. I would also like to thank Miaorun Wang, Johanna Blossei, Tilo Hartwig and Luis Zambrano for their continuous support during fieldwork, to Laurenz Teuber and Christian Schmidt for data transfer and to Evelyn Bolzmann for guidance on lab work. For short summary translation I am grateful to Weronika Matacz, Robert Strehse and Miarun Wang. For administrative support, I am grateful to Frau Karin Eckelt and Dr. Franziska Schmacka. Last but not the least, I would like to thank Dr. Martin Sperling and the entire Baltic Transcoast team of researchers and friends with I have spent long hours, either eating breakfast or being in Finland.

This doctoral dissertation was carried out as part of the WETSCAPES project (funded by the European Social Fund and the Ministry of Education, Science, and Culture of Mecklenburg-Western Pomerania [ESF/14-BM-A55-0028/16]).

Summary

Mires and peatlands play a large role in the global carbon cycle, covering only 3 percent of the world's terrestrial surface, but storing about 20 percent of the earth's total terrestrial carbon. However, over the last century, mires and peatlands have undergone severe degradation due to artificial drainage and deforestation to make way for agricultural exploitation. Nevertheless, the restoration of drained peatlands has gained much importance over the last three decades, mostly due to a better recognition of the multiple ecosystem services they provide such as carbon storage, habitat provision, and water flow regulation. As wetlands, mires and peatlands are heavily dependent on high water tables, as water-saturated conditions make it favorable for peat formation. Furthermore, peat-forming species of plants and mosses, depend on water tables at or near the surface. As such, site hydrology is vital for the maintenance of near-natural fens and the hydrological restoration (eg. rewetting) of drained fens is a pre-requisite for ecosystem recovery. Hydrological processes are closely linked to soil hydrophysical properties and as such investigations of both water dynamics and soil properties are vital for understanding ecosystem restoration success. Additionally, the effects of meteorological variables such as rainfall and temperature as controls on water table dynamics need to be assessed, especially in the face of recent European droughts and anticipated peatland vulnerability from climate change. The simplest restoration measure in place is rewetting either by blocking drainage ditches for inland peatlands or removing dikes for coastal peatlands to allow the re-establishment of hydrological connectivity with the sea. Given that the hydrology of minerotrophic peatlands (fens) remains understudied, especially in the temperate regions of mainland Europe, this dissertation aims to unravel the relationship of different environmental factors (environmental controls) with water-table dynamics in temperate fen peatlands to better inform sustainable management decisions. The specific research objectives are (1) to evaluate how long-term rewetting of drained fen peatland alters the response of the water table to precipitation (2) to quantify how such rewetting measures change the way meteorological factors (such as air temperature and relative humidity) drive water table dynamics and (3) to investigate whether soil surface microtopography controls hydrophysical properties of peat which has implications for overall hydrological processes in peatlands.

In a pair of percolation fens (1 rewetted and 1 drained site), multiple regression analysis between the rate of water table response to precipitation and precipitation intensity revealed that groundwater table in the rewetted fen has more than two times lower rate of response to precipitation events of a given intensity, compared to that of the drained fen, even after statistically adjusting for antecedent groundwater levels. Thus, the rewetted fen delivers a better hydrological buffer function against heavy precipitation events. It was found that for the depths at which the groundwater interacts with incoming precipitation, the peat of the rewetted fen has a higher specific yield (higher water storage capacity) causing groundwater to rise slower compared to the response at the drained fen. A period of 20 years of rewetting was sufficient to form a new layer of organic material with a substantial fraction of macropores providing storage capacity. Long-term rewetting has the potential to create favorable conditions (decreasing decomposition rates) for new peat accumulation, thereby altering water table response.

Within the same pair of percolation fens, a different approach (multiple regression on time series, adjusted for seasonal effects), revealed that meteorological factors such as temperature, precipitation and relative humidity control water table dynamics. It was found that a 1-degree

rise in daily maximum air temperature causes a drop of about 4 mm in the water table in the drained and degraded fen but only a drop of around 2 mm at the rewetted fen, principally through evapotranspiration. This was the case, even though the rewetted site showed almost two times higher dry-day evapotranspiration compared to the drained site. Higher minimum relative humidity limits evapotranspiration and thus causes a rise in the water table at both sites. Precipitation contributes to recharge, causing the water table to rise almost six times higher at the drained site than at the rewetted site. The differential impacts of meteorological conditions on water table dynamics can be attributed to (1) a difference in soil properties and (2) a difference in vegetation which act as surface controls.

In a rewetted coastal flood mire (non-tidal) located at the Baltic sea coast, it was found that the peat in the upper horizon with its very low saturated hydraulic conductivity (K_s) which is two or three orders of magnitude smaller than the underlying mineral layer, acts as a hydrological barrier to infiltration. Analysis of variograms revealed that soil organic matter content (SOM), K_s , and soil surface microtopography are all spatially auto-correlated within 100, 87, and 53 m. Bivariate Moran's I revealed a positive but weak spatial correlation between SOM and K_s and a moderately strong negative spatial correlation between SOM and soil surface microtopography. A map of soil organic matter content, generated by using simple Kriging method, predicts higher SOM in the center of the ecosystem, at lower elevations and lower SOM at the edges of the study area, at higher elevations. Local depressions in the center of the ecosystem provide a wetter and therefore more anaerobic environment, thereby decreasing carbon mineralization rates and enabling peat accumulation. The low hydraulic conductivity of the degraded peat along with the presence of lower micro-elevations in the center of the ecosystem is likely to increase the residence time of coastal floodwaters and thus may enhance (new) peat accumulation.

This cumulative dissertation underlines the importance of management regime (eg. long-rewetting), meteorological factors, and surface controls (eg. vegetation, microtopography, soil properties) for peatland restoration. Continuous monitoring of water-table and vegetation development in rewetted fen peatlands is advisable to ensure long-term success especially under climate change conditions and associated drought and dry spell events.

1. Introduction

1.1. Background

Mires and peatlands have a large role in the global carbon cycle (Zhong et al., 2020). They are high carbon ecosystems and depending on how they are managed, can be sources or sinks of greenhouse gases (Günther et al., 2020; Joosten et al., 2016; Rydin & Jeglum, 2013). Although peatlands cover only 3 percent of the world's land surface, they store approximately 20 percent of the global total soil organic carbon stock (Leifeld et al., 2019; Leifeld & Menichetti, 2018; Scharlemann et al., 2014) or two times that of global forest biomass (Humpenoeder et al., 2020). Other than their role in the carbon cycle, peatlands provide multiple ecosystem services (often with trade-offs). These include provisioning services (eg. Fuel, fiber, food, freshwater), regulating services (eg. climate and water regulation, water purification, erosion protection), cultural services (eg. recreational and educational services), as well as supporting services such as biodiversity, soil formation, and nutrient cycling (Bonn et al., 2016; Kimmel & Mander, 2010). However, over the last century, mires and peatlands have undergone extensive degradation due to artificial drainage to make way for agriculture and forestry. Other than drainage, peatlands face additional vulnerability from climate change (Hugelius et al., 2020; Qiu et al., 2020) with massive carbon banks being responsive to prolonged warming (Hopple et al., 2020). Much of the widespread drying of the European peatlands have been additionally attributed to climatic drivers (Swindles et al., 2019).

Between 1850 and 2015 temperate and boreal regions lost about 27 million ha and tropical regions 25 million ha of natural peatlands (Leifeld et al., 2019). It has been estimated that about 10 percent of global peatlands have been altered from long-term carbon sinks into sources (Joosten, 2010) for agriculture and forestry. In the 1960s, the global peatland biome changed from a net sink to a net source of greenhouse gas derived from the soil (Leifeld et al., 2019). If only Northern Peatlands are considered, at present, they have been found to have a net cooling effect on the climate. While it is uncertain whether they will continue to function as a net sink (Gallego-Sala et al., 2018), continued permafrost thaw may turn them into net sources of carbon (Hugelius et al., 2020). In terms of hydrological processes, peatland evapotranspiration has been shown to substantially exceed forest ET across the boreal region, and as the climate warms further, an increase in these differences is expected (Helbig, Waddington, Alekseychik, Amiro, Aurela, Barr, Black, Blanken et al., 2020).

Over the last two decades, there have been increasing efforts to restore the hydrological functionalities of drained and degraded peatlands (Menberu et al., 2016). Rewetting of drained peatlands either by ditch-blocking (for non-coastal peatlands) or by removal of dikes (for coastal fens) is an effective measure towards restoring the pre-existing hydrological regime (high water tables), which is crucial for peat-forming vegetation and peat accumulation (Lamers et al., 2015). As such peatland protection and restoration have been recognized as vital measures for climate change mitigation (Humpenoeder et al., 2020; Intergovernmental Panel on Climate Change, 2019; Leifeld & Menichetti, 2018) as well as "nature-based solutions" to achieve key EU sustainability objectives (Tanneberger, Appulo et al., 2020). Additionally, boreal peatlands, have recently been recognized for their *biophysical* climate mitigation (cooling effect) potential during the growing season compared for forests (Helbig, Waddington, Alekseychik, Amiro, Aurela, Barr, Black, Carey et al., 2020). Although rewetting may lead to methane emissions, prompt rewetting of drained peatlands reduces overall climate warming through the avoidance of CO_2 emissions (Günther et al., 2020; Nugent et al., 2019). The water table of a peatland is of pivotal importance as it drives ecosystem changes through changes to its biotic and abiotic components.

1.2. Peatland hydrology: Basic concepts

Peat is decaying organic matter that accumulates under water-saturated conditions. Peat formation thus occurs in areas where hydrological inputs exceed hydrological outputs and where there is a slow rate of decomposition. Peatlands Classification of peatland types is generally done according to two important factors which are sources of nutrients and sources of water. Bogs are ombrotrophic peatlands and are dependent on precipitation for water and nutrient supply whereas fens are minerotrophic peatlands and rely on groundwater for water and nutrients. As a result, Bogs are highly acidic with a pH less than 4 and are low in calcium and magnesium. Fens are less acidic and tend to be base-rich (Holden, 2006). There are several other ways of classifying mires and peatlands as well as sub-classifying bogs and fens which have been elaborated by Joosten et al. (2017). It is helpful to understand the different inputs and outputs and storage components in a peatland.

In terms of hydrological inputs precipitation is a major pathway through which recharge occurs. It is a general rule of thumb that bogs occur only in areas where there is over 600 mm of annual precipitation. In many blanket bogs in the Northern Hemisphere precipitation over 1500 mm is required depending on evapotranspiration rates as well as topography. However, fens may require less rainfall as they are dominated by groundwater. Groundwater inflow thus is another input to the water balance of a peatland. Subsurface springs may deliver water to fen complexes. Flooding in addition may be a source of water for valley floodplains. In tropical and temperate climate, peatlands are usually dependent on more than one source for water, while in arid regions, groundwater is the only substantial water component (Holden, 2006).

In terms of stores, water level data are the simplest to collect and usually the only data available. Peatlands in their natural state have water levels close to the surface. Thus, they have excess water that influences soil properties and the occurrence and distribution of flora. Peatlands function as a large store of water. Even above the water table, peat can hold a large volume of water (in the unsaturated zone). Even a small quantity of rainfall can be sufficient to raise the water table. Rainfall intensity is the dominant control upon the rate of rise. Variations in the water table elevation can substantially alter the ground surface level in peatlands. This is a particular issue in drained peatlands (Holden, 2006). In many peatlands, water is stored on the peat surface in what is known as peat pools (Whitfield et al., 2009). These pools form intricate networks ranging in size from localized systems to large complexes covering regional landscapes (Glaser, 1999).

In terms of losses, evapotranspiration is the principal way in which water is lost peatlands (Baird et al., 2004; Malloy & Price, 2014). At the scale of the ecosystem, the energy, water, and gas exchange processes are coupled strongly (Petrone et al., 2003). The evapotranspiration will be further discussed in section 1.4. Another component of water loss is runoff which includes both surface as well as subsurface flow. Peatland hydrological function within a catchment may vary over time and depend on the depth of the water table. Once a run-off threshold is crossed, there can be a change in a peatland's hydrological function – from transmission to run-off generation (Goodbrand et al., 2019).

One of the first experiments to examine the effects of drainage on the hydrological response of peatland catchments were Conway and Millar (1960). They reported results from four small peat catchments and concluded that peat drainage or burning causes increased sensitivity of runoff response to storm rainfall with higher and earlier peak flows. However, a study by Burke (1967) found that runoff was quicker from the undrained part of the bog with the water table very close to the surface while in the drained bog the water table was about half-a-meter deep causing runoff from the catchment to be slower. Similar results were reported by Baden and Eggelsmann (1968). McDonald (1973), stated that the results from Conway and Millar and Burke, while seemed to be in direct contrast, were not in fact so, due to the lack of comparability. The peat studied by Burke (1967) was more Sphagnum-rich and therefore had higher hydraulic conductivities than that of the more decomposed peat studied by Conway and Millar (1960). Thus McDonald concluded that drainage of one peat type will have a different effect on the runoff-rainfall relationship than drainage of another peat type. Another big difference between the sites, as Robinson (1980) and Robinson (1985) pointed out, was the drainage depths and densities which are very important for runoff generation. Holden et al. (2004, Table 1) summarize the differential hydrological effects of peatland drainage found by different studies on temporary storage, flood peak and annual runoff along with the catchment size, different processes they measured and discussed and point out how effects are complex and depend on local site conditions.

1.3. Water-table as the master variable for peatland restoration

The long-term development of peatland ecosystems is regulated by a complex network of interacting feedbacks among plant ecology, soil biogeochemistry, and groundwater and soil hydrology (Morris et al., 2011). Just as the natural flow regime of a river is the master variable for river systems (Sofi et al., 2020), water-table can be considered to be the master variable for peatland functionality. Peatlands are sensitive to hydrological changes that result from land-use changes or climate change (Holden et al., 2004). Even small alterations to the water table can substantially affect peatland biodiversity (Maltby, 1997). The climate sensitivity of methane emissions from northern peatlands have been shown to be mediated by seasonal hydrologic dynamics (Feng et al., 2020). Hydrological properties such as available water capacity characteristics are important for the survival of peat-forming species and strongly depend on the depth of the water-table (Price, 2003).

The substrate in peatlands is the peat, which consists of semi-decomposed plant material (Lindsay & Andersen, 2016). In other terms, peat is soil containing more than 30 percent organic matter (on a dry mass basis, Joosten & Clarke, 2002). Peat formation occurs when decomposition rates are lower than primary production (Moore, 1989). Organic matter needs large quantities of oxygen for aerobic decomposition to take place (Lindsay & Andersen, 2016). In natural or re-wetted peatlands, the decomposition rate is very slow, as the water table is at or near the surface (waterlogging conditions). As oxygen diffuses through water 10,000 times slower than it does through air, within the waterlogged mass of decomposing plant material, the supply of oxygen gets rapidly depleted (Clymo, 1983). Anaerobic decomposition is much slower than aerobic decomposition and in the presence of a steady supply of fresh organic material produced by peat-forming plants, there is an accumulation of the peat (Lindsay & Andersen, 2016). Maintaining high water tables is not only important for limiting decomposition and consequent soil carbon mineralization, but also for the survival of peat-forming plants (Timmermann et al., 2006). Without a constant supply of organic material into the resource pool, peat formation will halt quickly.

The maintenance of high water tables in peatlands is important for other reasons as well. By observing the different processes which occur following artificial drainage, the role of water tables in peatland functionality can be better understood. As mentioned earlier, water-saturated conditions prevent high rates of decomposition and enable a peat accumulating system and preserving peat depth (Schulte et al., 2019). Conversely, lowering the water table, for example by drainage, exposes the peat to an oxygen-rich environment. As such, aerobic decomposition occurs at a rapid pace, degrading the peat and starting a cascade of different hydrophysical and biogeochemical processes. As aerobic decomposition continues, carbon dioxide and nitrous oxide are released from the system further contributing to global climate forcing (Liu et al., 2019; Liu, Wrage-Mönnig et al., 2020). The carbon mineralization process causes the peat to become more consolidated and compacted, and as such the hydrophysical properties of peat change. In the absence of high water tables, the peat shrinks and land subsidence occurs, with a surface elevation declining from 0.5 to 4 meters over 50 years (Hooijer et al., 2012; Pronger et al., 2014). With increased effective stress, larger pores in the peat structure are the first to collapse as they are the least supported (Strack et al., 2008) and as such macroporosity decreases. Larger pores or macropores are responsible for transmitting most flow (Baird, 1997), and thus a reduction in macroporosity decreases saturated hydraulic conductivity (Whittington & Price, 2006). Changes to hydraulic properties of peat can affect the hydrology of peatlands (Whittington & Price, 2006)

As porosity is related to bulk density, a decrease in porosity increases bulk density. The reduction in pore size associated with greater bulk density has implications for water storage changes since peat that is more densely packed has lower specific yield and thus lower water storage capacity (Liu & Lennartz, 2019a; Liu, Price et al., 2020). Changes in such hydrophysical properties following drainage can increase water table fluctuations at a given site (Menberu et al., 2016; Whittington & Price, 2006). An increase in water-table fluctuation and can provide a positive feedback loop that could intensify further peat degradation, altering the dynamics of carbon cycling (Whittington & Price, 2006). The depth of the water table also determines the vulnerability of the peatland to peat fires and thus can provide further feedback into the climate system (Kettridge et al., 2015; Turetsky et al., 2015). The water table in peatlands has been shown to control chemical dynamics as well such as dissolved organic carbon transport (Strack et al., 2008; Webster & McLaughlin, 2010). The chemical composition of the peat substrate can also be influenced by even subtle changes to long-term water table position (Hribljan, 2012).

The different pathways through which changes occur in a peatland elaborates why the water-table can be considered as the master variable. While active changes to the management regime such as rewetting drained areas, can bring about desired changes for peatland restoration, other factors can have a large influence on water table dynamics. For the purpose of this dissertation, I will refer to the drivers as environmental controls. For the sustainable restoration and management of drained peatlands, a better understanding of the different drivers of water-table dynamics and associated processes (eg: evapotranspiration) is vital.

1.4. What drives water table dynamics and related hydrological processes?

While the water-table dynamics in a peatland drives several different ecosystem processes directly or indirectly, the water-table itself is driven by different biotic and abiotic factors. These may include environmental controls such as meteorological conditions, hydrogeological setting (Bourgault et al., 2019), microtopography, vegetation (Malhotra et al., 2016; Shi et al., 2015), local microclimatic conditions, soil properties (Taufik et al., 2019) amongst others.

Precipitation is a major source of groundwater recharge in peatlands. Incident rainfall infiltrates the peat surface and ultimately replenished the water table. As recharge occurs, the water table rises as a response. The magnitude of response of the water table to rainfall depends on several factors. These include initial depth of the water-table, interception by vegetation, soil hydrophysical properties (eg: specific yield), landscape position, and surface topography (Ferone & Devito, 2004; Meyboom, 1966), and hydrogeological setting (Bourgault et al., 2019), among others.

While precipitation is the primary way in which groundwater recharge takes place, evapotranspiration is the primary way through which discharge occurs in low-lying wetlands (Malloy & Price, 2014). Evapotranspiration is a complex ecohydrological process and is the product of evaporation from soil and water intercepted by the canopy, and transpiration from leaves (Ramirez & Harmsen, 2011). If the water storage capacity remains constant, higher evapotranspiration results in the lowering of the water table. The water table level has been found sometimes to be controlled almost entirely by evapotranspiration, especially during dry periods (Holden, 2006). Different meteorological factors affect evapotranspiration (Allen et al., 1998) and hence the water table. Higher temperature can lead to higher rates of evaporation because as temperature increases, the amount of energy required for evaporation decreases. Transpiration rates also increase with increasing temperatures in the absence of water stress (Will et al., 2013). Higher humidity decreases both evaporation and transpiration as it is easier for water to evaporate into dryer air than to wetter air (Ahmad, Hörmann et al., 2020; Shaw et al., 2010). Higher wind can facilitate evaporation and transpiration as the air around will be moved and more saturated air within proximity will be replaced by dryer air (Allen et al., 1998).

The microtopography of an ecosystem can also influence the water table in a wetland (Bruland & Richardson, 2005; Cresto-Aleina et al., 2015; Malhotra et al., 2016). This can include vegetational microforms such as hummocks, flats or hollows, or even small scale variations in the soil surface elevation, often consisting of local depressions and mounds. Microtopography development in peatlands includes feedbacks with vegetation and the water table depth (Malhotra et al., 2016). Varying peat accumulation rates of different species and species distribution is maintained by spatial differences in the moisture regime which in turn is related to the water table depth (see **Figure 1.1**).

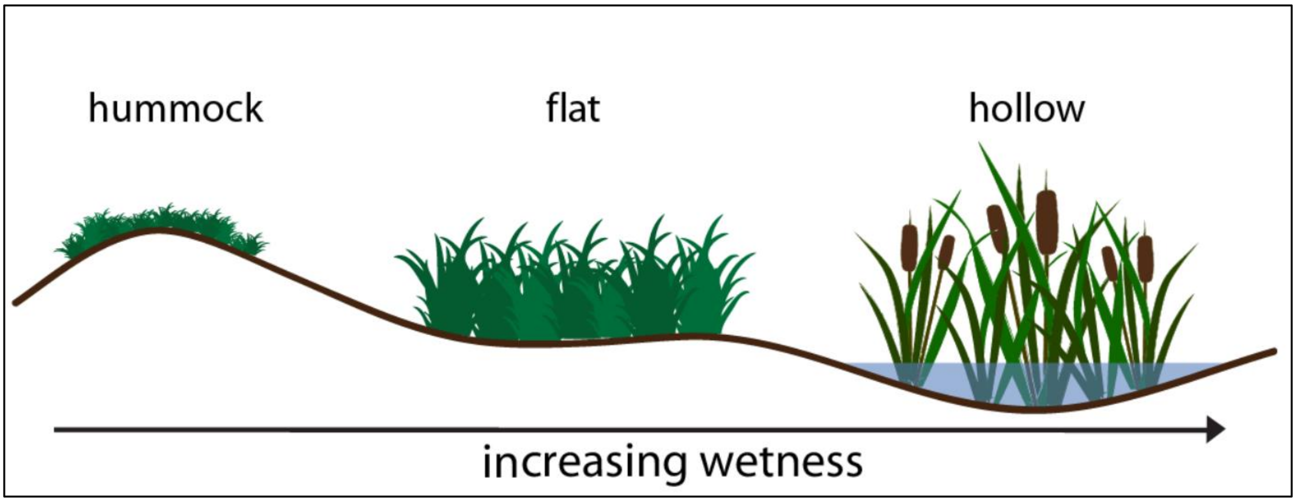


Figure 1.1 The association between microtopography and wetness in wetland soils. Adapted from Bruland and Richardson (2005).

While high water tables can influence soil properties, the reverse is also true. Soils such as peat with high soil organic matter can store a large amount of water, because of high pore volume, which also means that effective porosity (also known as specific yield) is high. The high water storage capacity of peat soil means that even if a large volume of water is removed from or added to the system, the water level will not rise or fall as much as it would in soils with low water holding capacity. As such, soil hydrophysical properties is an important factor influencing water table dynamics in peatlands.

1.5. Hydrology of fen as an understudied area

While there have been numerous studies in wetland hydrology, efforts to understand the hydrology of mires and peatlands have increased only slowly over the last few decades. Additionally, most hydrology studies have focused on bogs rather than fens. Even those which have focused on fens have done so for boreal fen peatlands, rather than temperate ones. As such, much remains to be understood regarding hydrological processes in temperate fen peatlands (see **Figure 1.2**).

According to a bibliometric analysis by van Bellen and Larivière (2020), in the early 1990s about half of the articles on peatlands were based on boreal-arctic sites. According to them, hydrology research was highly concentrated geographically, with about 46 percent of the sites located in Canada and the USA, with the UK accounting for 14 percent. A literature search in SCOPUS with the terms (1)“peatland” AND “bog” and “hydrology” OR “peatland” AND “bog” and “water” (2) “peatland” AND “fen” and “hydrology” OR “peatland” AND “fen” and “water” reveals that fen hydrological research has only recently caught up with the numbers of studies published in bog hydrology.

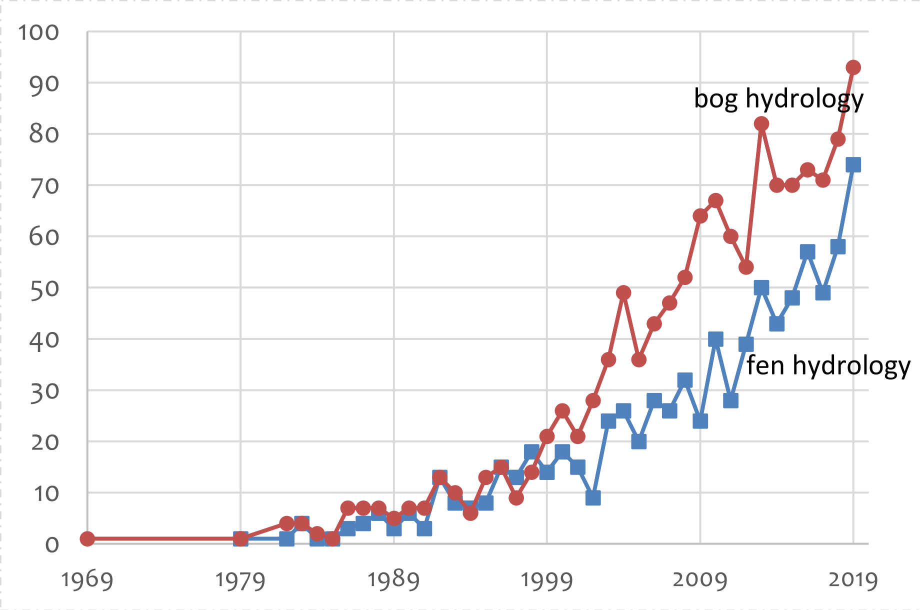


Figure 1.2 Trends in peatland hydrological research.

More specifically, the effect of long-term (>20 years) rewetting on fen peatland hydrology and how such changes in management regime may modulate the different environmental controls on the water table is lacking, mostly due to lack of data.

1.6. General research objectives and approach

The general research objective of this study is to unravel the relationship of different environmental factors (environmental controls) with water-table dynamics in temperate fen peatlands to better inform sustainable management decisions. The specific research objectives are (1) to evaluate how long-term rewetting of drained fen peatland alters the response of the water table to precipitation (2) to quantify how such rewetting measures change the way meteorological factors (such as air temperature and relative humidity) drive water table dynamics and (3) to investigate whether soil surface microtopography controls hydrophysical properties of peat which has implications for overall hydrological processes in peatlands. The studies which address (1) and (2) were carried out in two inland fens (1 drained and 1 rewetted), while the research which addresses (3) was carried out in a rewetted coastal fen. This study has been conducted under the interdisciplinary project WETSCAPES, which aims to develop scientific principles for sustainable management of fen peatlands, particularly those which underwent drainage and was later rewetted. The WETSCAPES approach has been described in detail by Jurasinski et al. (2020).

1.7. Structure of this dissertation

This work is a cumulative dissertation and is divided into 5 chapters. **Chapter One** (the current chapter) sets the background against which the research work contained in this dissertation has been carried out. This includes introducing the reader to the importance of mires and peatlands for their role in ecosystem service delivery, giving a brief idea of why and how the water-table in a peatland is the main driver of ecosystem processes and environmental change, and what environmental controls drive water-table dynamics through different hydrological processes. The chapter also briefly describes the main aims and specific objectives.

Chapter Two investigates the effects of long-term rewetting on peatland water table dynamics. It evaluates how the response of the water-table to rainfall events relates to the hydrophysical properties of peat. This chapter has been published as an original research article titled "**Long-term rewetting of degraded peatlands restores hydrological buffer function**" in *Science of the Total Environment* (Ahmad, Liu, Günther et al., 2020).

Chapter Three analyzes how precipitation along with other meteorological factors such as relative humidity and air temperature act as controls over the water-table. It investigates how the existing management regime may modify the effects of such controls by comparing a rewetted fen with a drained fen. This chapter has been submitted to the Special Issue "Observing, modeling and understanding processes in natural and managed Peatlands" by *Frontier in Earth Science*, to be published as an original research article titled "**Meteorological controls on water table dynamics in fen peatlands depend on management regimes**" (Ahmad, Liu, Alam et al., 2020 ; under review).

Chapter Four presents a spatial analysis of soil properties such as saturated hydraulic conductivity and soil organic matter and elucidates how soil surface microtopography is associated with such properties a coastal fen. Heterogeneity of soil properties is presented along with how they are spatially auto-correlated. This chapter has been published as an original research article in *Mires and Peat* with the title "**Spatial heterogeneity of soil properties in relation to microtopography in a non-tidal rewetted coastal mire**" (Ahmad, Liu, Beyer et al., 2020).

Chapter Five provides a concluding discussion of the overall dissertation. Through a composite schematic diagram, it synthesizes the novel contribution of the dissertation to a better understanding of fen peatland hydrology. It goes beyond the results and discussions of Chapters 2-4 and sheds light on a multitude of implications for fen peatland restoration and management.

The readers should note that to keep the coherence in the dissertation format, the original articles which make up Chapters Two and Four, and the submitted manuscript which makes up Chapter three, **have been reformatted**, in terms of the numbers of columns and the figure numbering and placement, and coherent spelling (American Standard English). The contents of these studies remain the same.

2. Long-term rewetting of degraded peatlands restores hydrological buffer function

Sate Ahmad [1]*, Haojie Liu [1], Anke Günther [2], John Couwenberg [3], Bernd Lennartz [1]

[1] Soil Physics, Faculty of Agricultural and Environmental Sciences, University of Rostock, Justus-von-Liebig-Weg 6, 18059 Rostock, Germany

[2] Landscape Ecology and Site Evaluation, Faculty of Agricultural and Environmental Sciences, University of Rostock, Justus-von-Liebig-Weg 6, 18059 Rostock, Germany

[3] Peatland Studies and Palaeoecology, Institute of Botany and Landscape Ecology, University of Greifswald, partner in the Greifswald Mire Centre, Soldmannstraße 15, 17487 Greifswald, Germany

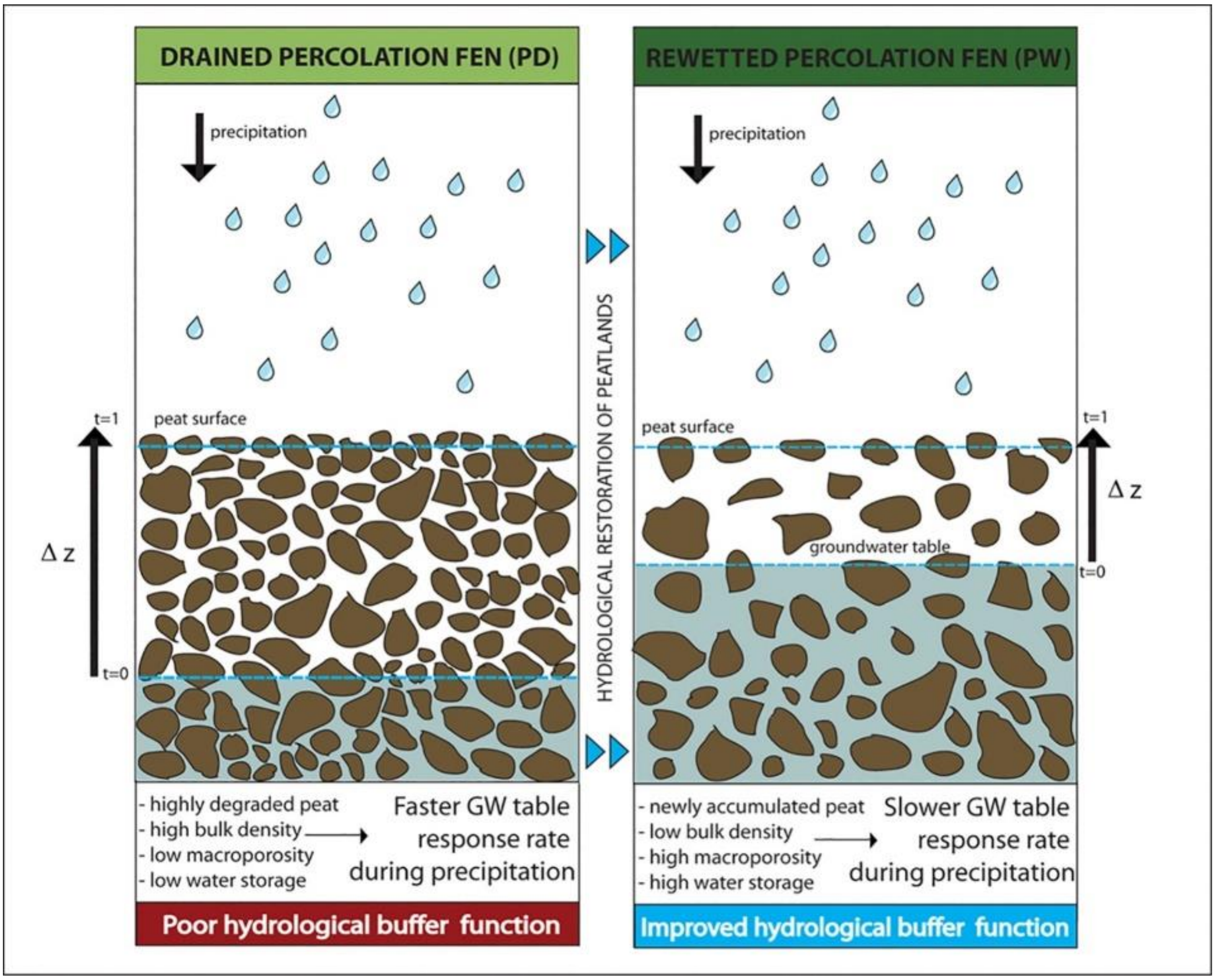
*Corresponding Author (~80% contribution)

Abstract

Precipitation is a key factor affecting shallow water table fluctuations. Although the literature on shallow aquifers is vast, groundwater response to precipitation in peatlands has received little attention so far. Characterizing groundwater response to precipitation events in differently managed peatlands can give insight into ecohydrological processes. In this study we determined the groundwater table response rate following precipitation events at a drained and a rewetted fen to characterize the effect of rewetting on hydrological buffer capacity. Multiple regression analysis revealed that the groundwater table at the rewetted fen has more than two times lower rate of response to precipitation events than that of the drained fen, even after adjusting for antecedent groundwater levels. Thus, the rewetted fen delivers a better hydrological buffer function against heavy precipitation events than the drained fen. We found that for the depths at which the groundwater interacts with incoming precipitation, the peat of the rewetted fen has a higher specific yield causing groundwater to rise slower compared to the response at the drained fen. A period of 20 years of rewetting was sufficient to form a new layer of organic material with a significant fraction of macropores providing storage capacity. Long-term rewetting has the potential to create favorable conditions for new peat accumulation, thereby altering water table response. Our study has implications for evaluating the success of restoration measures with respect to hydrological functions of percolation fens.

Keywords: Hydrological restoration; Groundwater response; Percolation fens; Soil physical properties

Graphical Abstract



Highlights

- Groundwater response to rainfall can be used to evaluate hydrological functioning.
- The rewetted fen responds at a slower rate to precipitation than the drained one.
- Long-term rewetting can restore water storage capacity of the upper soil horizon.

2.1. Introduction

Mire and peatland ecosystems play a significant role in the global carbon cycle as they are sources or sinks of greenhouse gases (GHG), depending on how they are managed (Günther et al., 2020; Joosten et al., 2016; Leifeld et al., 2019; Rydin & Jeglum, 2013). Although, peatlands cover a small portion of the Earth's land surface (around 3%), they store about 20% of the global total soil organic carbon stock (Joosten et al., 2016; Leifeld & Menichetti, 2018; Scharlemann et al., 2014). However, at present artificial drainage of about 10% of global peatlands have changed them from long-term carbon sinks into sources (Joosten, 2010). Conservation of high-carbon ecosystems such as peatlands will have immediate positive impact in terms of emission reductions (Intergovernmental Panel on Climate Change, 2019) and emission avoidance through peatland restoration has been recognized as climate change mitigation strategy (Leifeld & Menichetti, 2018; Wilson et al., 2016).

Peatlands may provide other ecosystem functions and services besides carbon storage, such as the hydrological buffer function (Lennartz & Liu, 2019). In a catchment or river basin, peatlands may transform heavy rainfall events into a smooth discharge curve depending on their hydraulic properties and connectivity. Crucial in this context is the specific yield, which is defined as the fraction of the peat volume that drains instantaneously as the water table descends and as such it determines the water storage capacity of the peat body (Heliotis, 1989).

Artificial drainage of peatlands causes the water table to drop and subsequent collapse and decomposition of the peat alters hydro-physical properties such as pore structure, hydraulic conductivity (Ivanov, 1981; Liu et al., 2016; Price, 2003; Rezanezhad et al., 2016; Zeitz & Velty, 2002) and soil organic matter content (Heller & Zeitz, 2012). Peatland drainage will also change the vegetation composition (Schrautzer et al., 2013), overall biodiversity (Maltby, 1997), water chemistry (Holden et al., 2004) and hydrological processes (Holden et al., 2006; Holden & Burt, 2003)). In peatlands, the survival of peat-forming plant species and the necessary high and stable water tables depend on hydrological properties of the peat (Menberu et al., 2016; Price, 2003).

In recent years there have been increasing efforts towards the restoration of hydrological functions of degraded peatlands by rewetting through blocking old drainage networks (Kotowski et al., 2016; Lamers et al., 2015), as rewetting can minimize further degradation or even create favorable hydrological conditions for the new formation of peat (Menberu et al., 2016; Mrotzek et al., 2020; Niedermeier & Robinson, 2007). However, to date the restoration of peatland hydrological functions is not sufficiently understood. Price et al.'s (2003) paper provides an overview of how hydrological processes differ in abandoned and in restored peatlands. A study by Menberu et al. (2016) evaluated the restoration of peatland hydrology in 24 different peatlands.

The degradation of peat subsequent to drainage decreases the hydraulic conductivity (Ivanov, 1981; Liu & Lennartz, 2019a; Price, 2003), which refers to the ability of soil to transmit water (Amoozegar & Warrick, 1986). A lower hydraulic conductivity would decrease the flow of water to and through the peat matrix, reducing the connectivity of a peatland to adjacent mineral soils and within the landscape as a whole. Likewise, macroporosity decreases dramatically upon drainage, lowering the ability of the peat body to store water (Liu & Lennartz, 2019a). Such changes in peat properties can affect the response of the water table to precipitation with possible consequences for discharge (Menberu et al., 2016).

Analysis of water table response to rainfall is an effective means to understand restoration success as it gives insight into physical properties of peat such as bulk density or even specific storage capacity of peatlands. While there have been numerous studies on water table response to precipitation conducted in shallow and deep aquifers (Cai & Ofterdinger, 2016; Zhang et al., 2017), such analyses have been limited in peatlands (Bourgault et al., 2019; Heliotis & DeWitt, 1987; Romanov, 1968; van der Schaaf, 1999). There have been several studies on water table dynamics and on the impact of restoration measures on-site hydrology in bogs (D'Acunha et al., 2018; Green et al., 2017; Holden et al., 2006; Holden et al., 2011; Howie et al., 2009; Ruseckas & Grigaliūnas, 2008). For example, the study by Holden et al. (2011), found that after 7 years of rewetting by ditch blocking in a blanket bog, the hydrological changes caused by 40 years of drainage were not entirely reversed. Nevertheless, they did find some evidence of a slow recovery as indicated by water table responses to rainfall events and seasonality of water table variability. However, there has been a lack of studies on the effects of rewetting on water table dynamics in temperate peatlands, especially in percolation fens.

Percolation fens are minerotrophic peatlands that depend on a large supply of water that is evenly distributed throughout the year. As such, they depend on large catchment areas and are commonly found in river valleys or as part of valley mire systems. Valley mire systems typically consist of spring mires at the edges, wide percolation mires and strips of flood mires along the course of the river (Succow & Joosten, 2001; Tiemeyer et al., 2006). In their natural state, percolation fens are characterized by a high and stable water table, which results in only slightly decomposed peat with high hydraulic conductivity causing water to percolate through the whole peat body (Joosten et al., 2017; Joosten & Clarke, 2002; Lindsay, 2016). The German federal state of Mecklenburg-Western Pomerania has the highest density of fens in Germany, accounting for 12% of the total land area, of which one third are percolation fens (Koch & Jurasinski, 2015; Zauft et al., 2010). These are mostly located along river valleys which are remains of melt water channels of the last glaciation, where permanent groundwater flow from adjacent moraines caused paludification (Koch & Jurasinski, 2015; Succow & Joosten, 2001), a process through which organic matter is accumulated over time resulting from increasing soil moisture and colonization of peat forming species (Lavoie et al., 2005).

The understanding of hydrological restoration of temperate peatlands through long-term (> 20 years) rewetting is limited. This is especially true for percolation fens. The objective of this study was to evaluate the restoration of hydrological functions in peatlands by comparing a rewetted (PW) and a drained percolation fen (PD). We aimed at determining the groundwater table response rate following precipitation events at both sites, taking into account the effect of antecedent groundwater levels. Furthermore, we wanted to test the predictive power of selected soil physical properties for restoration success and hydrological buffer capacity.

2.2. Materials and methods

2.2.1. Study sites

The two study sites are 8 km apart and are located in the Federal State of Mecklenburg-Vorpommern, Germany and together belong to the largest connected fen complex in northeastern Germany (Jurasinski et al., 2020). One site is a drained percolation fen (PD), while the other has been rewetted 23 years ago (PW), but was formerly drained for decades (**Figure 2.1**). They are typical percolation fens, which formed in meltwater channels left by the retreating ice after the last glacial period. PD is a fairly homogenous grassland, consisting mainly of *Ranunculus repens* L. and *Deschampsia cespitosa* (L.) P. Beauv. with some *Holcus lanatus* L. and *Poa trivialis* L., while PW is more diverse with a mosaic of several dominant stands that developed after rewetting. The studied plot is dominated by *Carex acutiformis* Ehrh., with few occurrences of *Epilobium hirsutum* L. PW was drained since c. 1750, and drainage was much intensified in c. 1970 for high intensity pasture management. In 1997, the site was rewetted as a part of the state peatland conservation program and the EU-LIFE program. Today, PW can be considered to be a near-natural percolation fen, part of a larger valley mire system (Tiemeyer et al., 2006). The relevant site characteristics have been summarized in **Figure 2.2**.

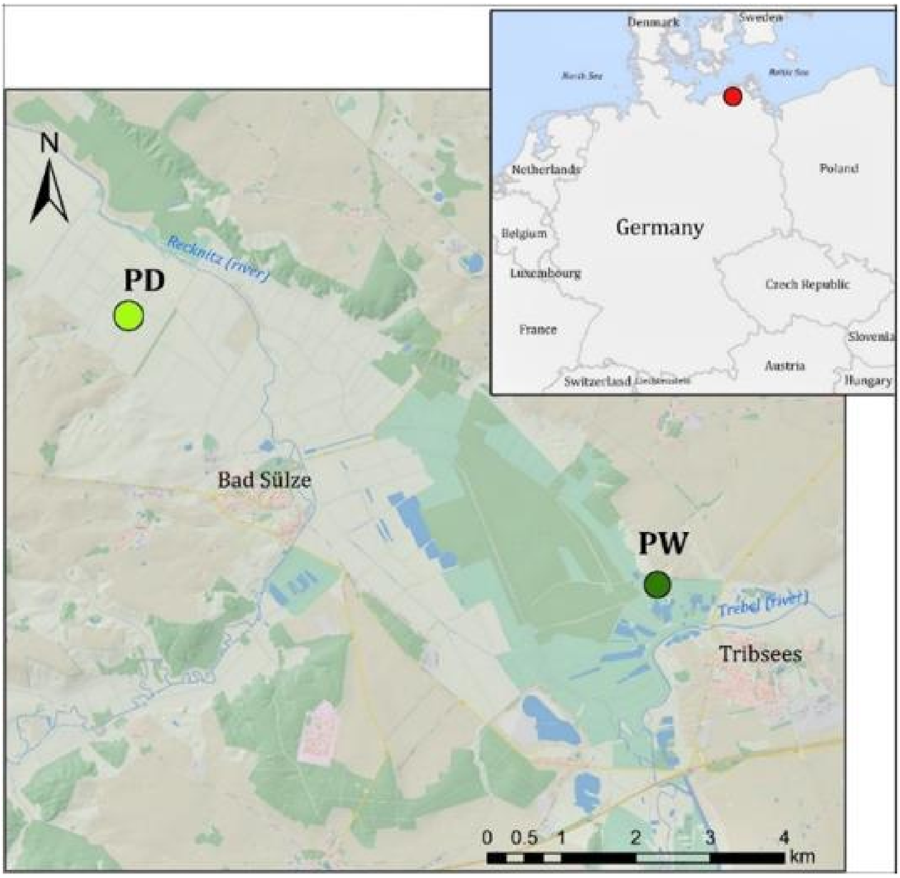
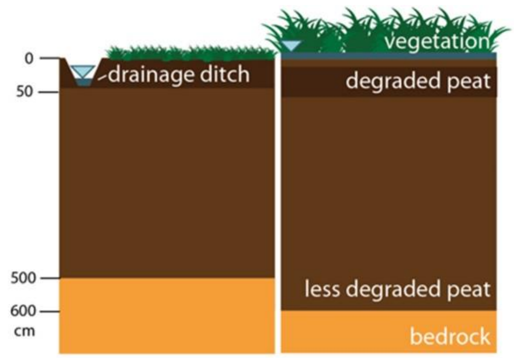


Figure 2.1 Location of the study sites: drained (PD) and rewetted (PW) percolation fens (Basemap: , 2020, Underlying relief: GeoPortal.MV, 2020). Red dot on inset map shows the study locations with respect to the national boundary of Germany (Country maps: Eurostat, 2020).



Site ID	PD	PW
State	drained	rewetted
Year drained	~1750 and 1970	
Year rewetted	-	1997
Vegetation	grassland	sedge reed
Peat depth	500 cm	600 cm
Dry bulk density (top soil)	0.28 g cm^{-3}	0.19 g cm^{-3}
Soil pH	5.3	5.4

Figure 2.2 Summary characteristics of the study sites, modified after Jurasinski et al. (2020).

2.2.2 Hydrological data registration and analysis

As part of the WETSCAPES project (Jurasinski et al., 2020), one water level logger (SEBA hydrometrie Dipper-PT) was installed for each site and data was registered every 15 min. Weather stations were also installed at each site, which registered precipitation (YOUNG Tipping Bucket Rain Gauge), among other meteorological variables, every 10 min.

For our analysis we used data from 30 September 2017 till 9 June 2019 for PW and till 24 July 2019 for PD. All data were merged together and temporal scales were matched to 30-minute intervals employing the R statistical computing software (R Core Team, 2019). A user-defined function was developed with which precipitation events and their respective duration were calculated using a moving temporal window of 6 h for which summation of precipitation and duration would halt if rainfall stopped for 30 min. Using cross-correlation functions, no significant lags were found between precipitation and water table rise within the time intervals for either site, indicating an immediate reaction. The response rate was calculated as the rise in water table divided by the time it took to rise. Precipitation intensity (rate of precipitation) was calculated by dividing the precipitation event size by the duration of the event. All calculations were done for time periods when the water level was below the soil surface. This is because when water table is above the surface, any response to rainfall would not reflect the physical properties of the peat. Bivariate relationships such as those between precipitation intensities and groundwater response rates were investigated using scatterplots and Pearson correlations. Multiple linear regression analyses were computed separately for PD and PW with response rate as the dependent variable and precipitation intensity and antecedent groundwater level as the independent variables.

2.3. Soil dry bulk density and specific yield determination

In order to corroborate the results from groundwater table response analysis, we dug out one undisturbed soil column (20 cm in diameter) per site in July (PW) and August of 2017 (PD) and stored them at −17 °C. The uppermost 43 cm (PD) and 55 cm (PW) of the soil column were cut into contiguous 0.5 cm slices at −4 °C using the DAMOCLES device (Joosten & Klerk, 2007). The slices were volumetrically subsampled for analyses of bulk density and organic matter content. Subsamples of 6 cm3 were weighed for bulk density and ignited at 550 °C to measure the loss of organic matter. For PD, since the water table drops beyond a depth of 43 cm, additional sampling of soil cores was carried out at the depths of 50, 70 and 100 cm to determine bulk density.

Bulk density data (first 2 cm not considered because of potential disturbance and thick roots) was converted to specific yield using a pedotransfer function (specific yield = 0.003 × bulk density-1.40) developed by Liu, Price et al. (2020) for peat soils (see **Figure 2.3**) **.** Specific yield is the fraction of the peat volume that drains instantaneously as the water table descends and as such it determines the water storage capacity of the peat body (Heliotis, 1989). Following Thompson and Waddington (2013), Liu, Price et al. (2020) calculated specific yield as the difference between total porosity and volumetric water content at −10 cm water pressure head. Note that values of specific yield are more sensitive to bulk density if bulk density is lower than 0.2 g cm^{-3}.

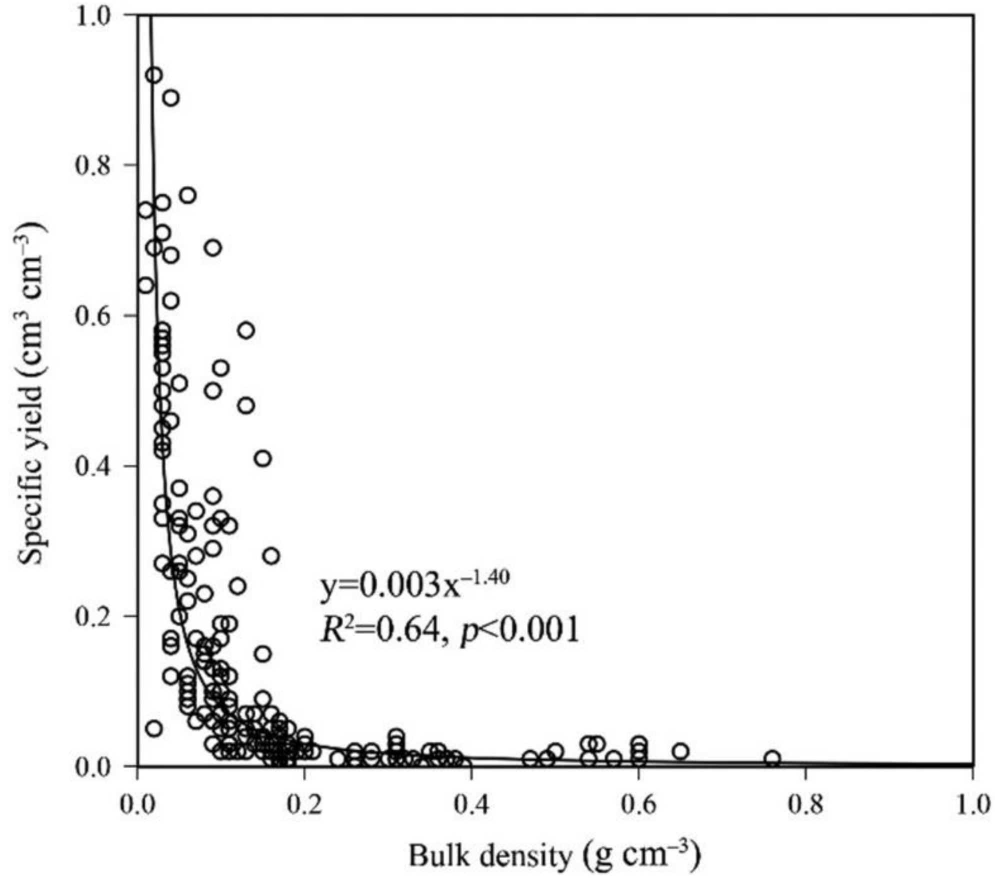


Figure 2.3 Pedotransfer function to convert bulk density of peat to specific yield (Liu, Price et al., 2020).

Additionally, the precipitation event size to groundwater response ratio (PPT/RSP) was calculated for each site and compared with the specific yield values derived from the pedotransfer function. PPT/RSP is a good indicator of specific storage capacity and has been used as a proxy for specific yield in several studies (Ahmad, Hörmann et al., 2020; Chin, 2008; Heliotis, 1989; Van Gaalen et al., 2013; Zhang et al., 2017), the central assumption being that all of the input from rainfall contributes to the water table rise. Yet, this assumption may not hold true in cases where the unsaturated zone retains a substantial volume of water. Therefore, the values of PPT/RSP are only an indication of specific yield.

A summary of the materials and methods used in the study is provided in **Table 2.1**

Table 2.1 Summary of the materials and methods used in the study.

Variable/data	Instrument/tool	Date	Description/use/analysis
Water level	SEBA hydrometrie Dipper –PT (one, per site)	30 September 2017 till 9 June 2019 (PW) and till 24 July 2019 for (PD)	Data was registered every 15 min. Response rate calculated rise in groundwater table during precipitation event divided by duration of the rise.
Rainfall	YOUNG Tipping Bucket Rain Gauge	30 September 2017 till 9 June 2019 (PW) and till 24 July 2019 for (PD)	Data registered every 10 min. Timesteps of water table and rainfall were matched to 30 min intervals. Precipitation event size calculated using a user defined function in R, by summing up precipitation for as long as there is rain. The moment rainfall stops, the summing stops marking the end of an event. Precipitation intensity was calculated as precipitation event size divided by duration
Soil samples	1 undisturbed soil column per site; DAMOCLES soil cutting apparatus (Joosten & Klerk, 2007).	July 2017 (PW); August 2017 (PD)	Undisturbed soil columns stored at −17 °C. Upper 42 cm (PD) and 55 cm (PW), were cut at 0.5 cm slices at −4 °C. The slices were subsampled volumetrically for analysis of bulk density.
Specific yield calculation	Pedotransfer function to transform peat bulk density to specific yield (Liu, Price et al., 2020)	–	Specific yield = 0.003 × bulk density$^{-1.40}$

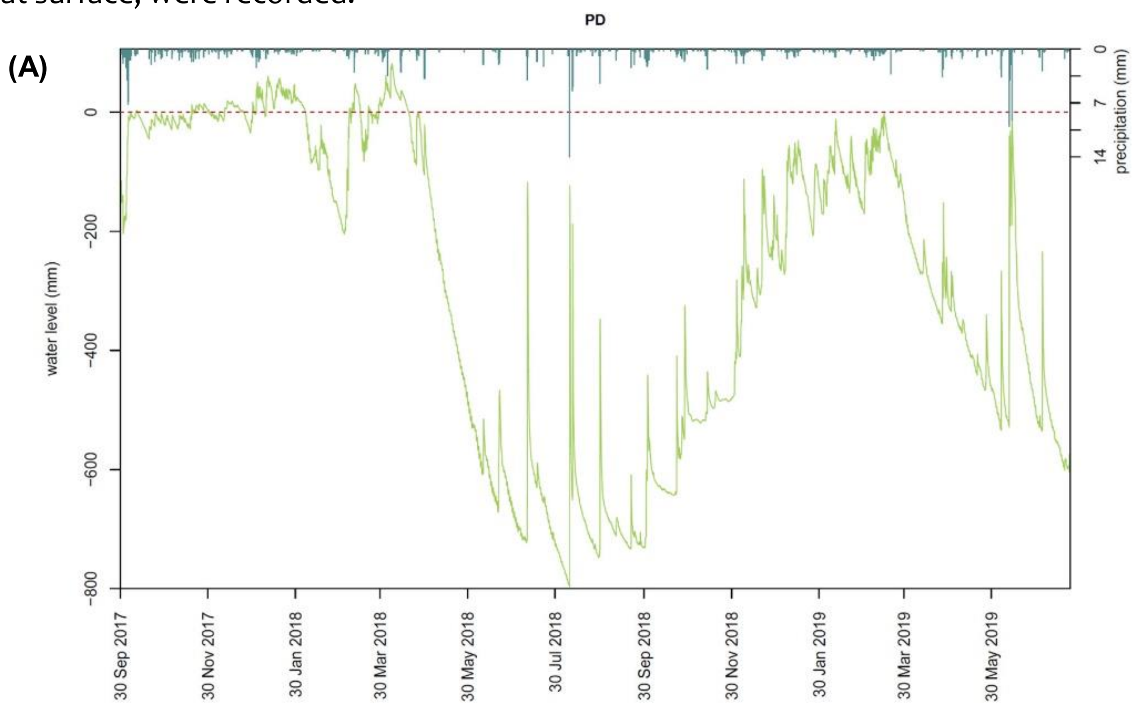
Table 2.2 Mean and confidence intervals [lower limit, upper limit] of different hydrological variables. '*N*' denotes the sample size, which is the number of precipitation events.

Site	Precipitation event size (mm)	Precipitation intensity (mm/h)	Water table response (mm)	Water table response rate (mm/h)	Antecedent groundwater level (mm)	*N*
PD	3.57 [2.90, 4.25]	0.27 [0.22, 0.32]	41.08 [29.69, 52.48]	3.36 [2.25, 4.48]	−281.4 [−318.7, −244.3]	181
PW	5.03 [2.70, 7.35]	0.39 [0.23, 0.56]	37.13 [23.11, 51.14]	3.07 [1.97, 4.16]	−155.7 [−179.1, −132.2]	51

2.3. Results

2.3.1. Hydrographs, water table characteristics and precipitation events

The annual average water table at PW for the year 2018 was −39 mm while it was −367 mm at PD. From the hydrographs of water table and precipitation (**Figure 2.4**), we can observe that water table responds immediately to precipitation in both sites. For both sites, seasonality of precipitation as well as groundwater table can be observed, with low water tables in spring and summer and high water tables in autumn and winter. Although PD is a drained site, the water table rises near or even over the surface several times during the year, especially at the end of 2017. This can be attributed to the exceptionally wet conditions of the year 2017 with a total annual precipitation anomaly of +187 mm, relative to the long-term average (1981 to 2010; Jurasinski et al., 2020). However, over the course of the following year, the groundwater table fell to a maximum depth of −800 mm in PD which can be attributed to the exceptionally dry year of 2018, with a total annual precipitation anomaly of −126 mm. Such a lack of precipitation input also caused a drop in the water level, even at PW, where water levels of up to 300 mm below the peat surface, were recorded.



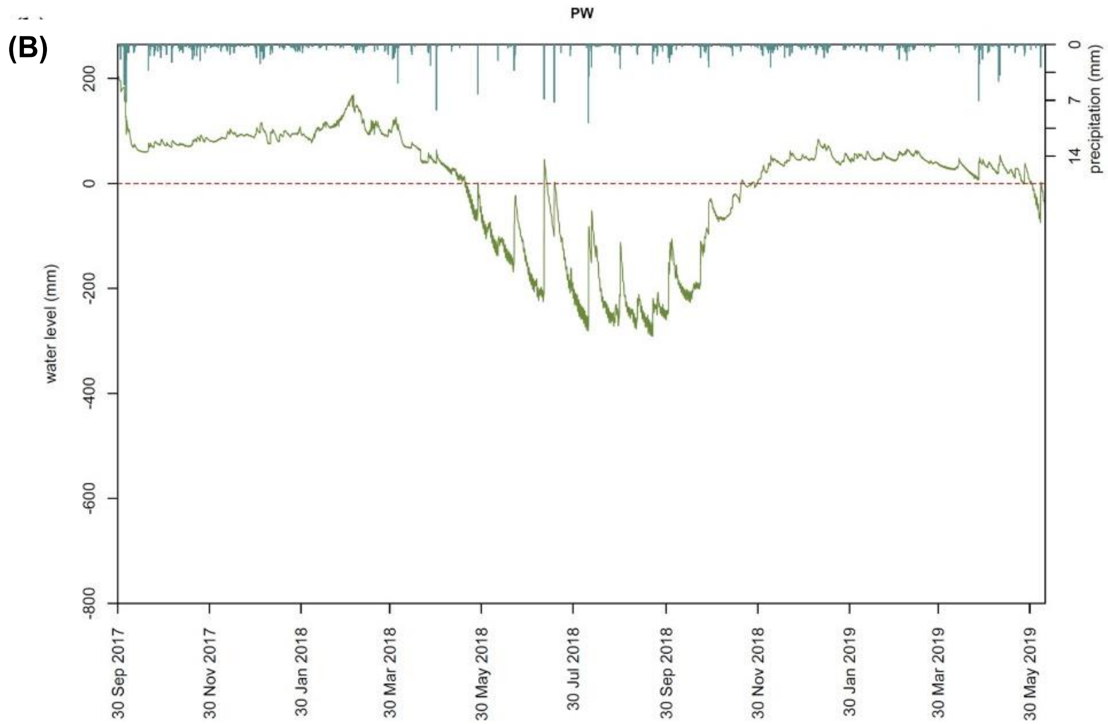


Figure 2.4 Water level (1st y-axis, green line) and precipitation (2nd y-axis, blue spikes) for **(A)** Drained (PD) and **(B)** Rewetted (PW) fens. The dotted red line indicates the peat surfaces. In total 181 and 51 precipitation events were identified for PD and PW, respectively. Summary statistics of groundwater level and precipitation are shown in **Table 2.2**.

2.3.2. Bivariate relationships between hydrological variables

Simple linear regression analysis between the groundwater table response and precipitation event size (**Figure SM 2.1**) reveals that for each unit increase of rainfall event size, groundwater table rises by about 12 mm (R^2 = 0.50) at PD. For PW, 1 mm increase in rainfall event size causes a half as large increase of about 6 mm groundwater table (R^2 = 0.92). The much lower R-squared value for PD shows that the response is less predictable than for PW.

Simple linear regression between groundwater response rate and precipitation intensity reveals significantly strong linear correlations for both PD (R^2 = 0.71) and PW (R^2 = 0.93). The response rate is much faster at PD than at PW (**Figure 2.5**). At PD, each unit increase in precipitation intensity results in a corresponding increase of 17.9 units in response rate; at PW, the corresponding increase in groundwater response rate is only 6.3 units. The difference in response can be related to a difference in hydrophysical properties of the peat. The water table immediately before the start of the precipitation event, which we refer to as antecedent groundwater level, should be taken into account using multiple linear regression models.

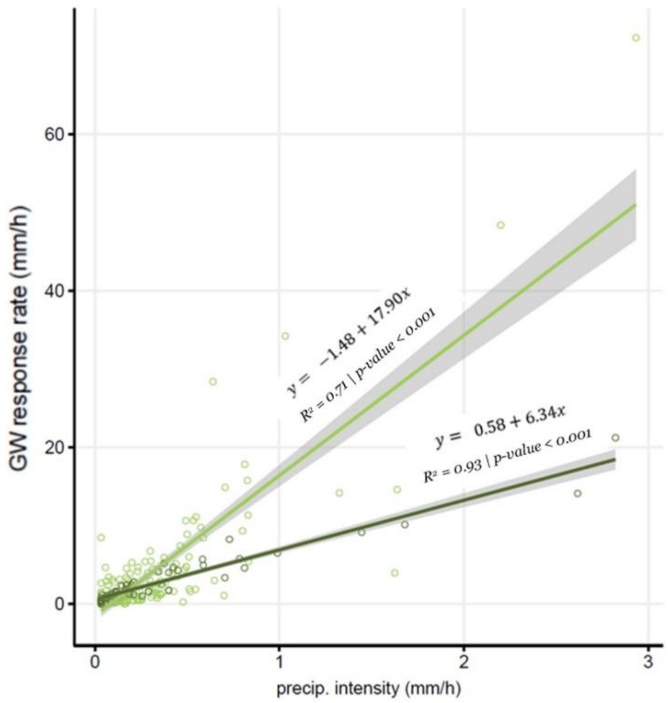


Figure 2.5 Groundwater level (GW) response rate and precipitation event intensity at drained (PD, light green) and rewetted (PW, dark green) fen. Shaded areas represent the 95% confidence intervals.

There is a rather weak linear relationship between the antecedent groundwater level and the response rate for both sites (**Figure SM 2.2**). Contrary to our expectation that the response rate would be more pronounced at shallower depths, it appears to be marginally higher at deeper depths.

2.3.3. Water table response to combined effects of precipitation intensity and antecedent groundwater level

After accounting for antecedent groundwater level, every unit increase in precipitation intensity increases the rate of groundwater response by 6 units at PW and by 17 units at PD (**Table 2.3**). Thus, the groundwater table response at PD is more than two times more sensitive to precipitation intensity than at PW.

Table 2.3 Results of multiple regressions with groundwater response rate (mm/h) as the dependent variable at PW and PD.

Sites	Regression terms	Estimate	Std. error	t value	Pr(>|t|)
	Intercept	−0.365	0.287	−1.271	0.210
PW	Antecedent water table (mm)	−0.006	0.002	−3.967	<0.001***
	Precipitation intensity (mm/h)	6.228	0.227	27.448	<0.001***

Residual standard error: 0.942 on 48 degrees of freedom; Multiple R-squared: 0.944|Adjusted R-squared: 0.942|F-statistic: 404 on 2 and 48 DF|p-value: < 0.001.

Sites	Regression terms	Estimate	Std. error	t value	Pr(>|t|)
PD	Intercept	−2.623	0.473	−5.548	<0.001***
	Antecedent water table (mm)	−0.005	0.001	−3.868	<0.001***
	Precipitation intensity (mm/h)	17.389	0.832	20.895	<0.001***

Residual standard error: 3.958 on 178 degrees of freedom; Multiple R-squared: 0.730|Adjusted R-squared: 0.733|F-statistic: 244.8 on 2 and 178 DF|p-value: < 0.001.

****p < 0.001.*

2.3.4. Physical properties of peat

At PD bulk density shows a sharp increase from a minimum of 0.04 g cm−3 at 2.5 cm depth to a maximum of 0.23 g cm−3 at a depth of only 6.5 cm. Below 6.5 cm depth the bulk density steadily decreases with depth. In contrast, at PW, BD values reaches as high as the maximum BD of PD at a much larger depth, around 20 cm below the peat surface. However, bulk density at PW peaks off at 0.35 g cm−3, at a depth of 27 cm, then starts to decrease, first sharply and then more gradually (**Figure 2.6**).

Applying the pedotransfer function, both sites show sharply decreasing estimated specific yield (SY) down to a certain depth, after which there is gradual increase. In the topsoil, PW has much higher SY than PD: The mean SY at 0–10 cm depth is 0.06 for PD and 0.21 for PW, and for 10–20 cm depth 0.04 for PD and 0.07 for PW (**Table 2.4**). For the depth range of water table fluctuations (upper 800 mm for PD and upper 300 mm for PW), PD has significantly lower SY than PW (mean ± SEM: 0.06 ± 0.00 vs. mean ± SEM: 0.09 ± 0.01; tested with Welch-corrected two-sample *t*-test, p-value < 0.01). Note that the values of specific yield are more sensitive to bulk density when it is below 0.2 g cm−3.

The precipitation event size to water level response ratio (PPT/RSP) is a good indicator of specific storage capacity, but has also been used as a proxy for specific yield in several studies (Ahmad et al., 2020; Chin, 2008; Heliotis, 1989; Van Gaalen et al., 2013; Zhang et al., 2017). The PPT/RSP ratio over depth shows a good agreement with specific yield, as derived from the bulk density data, especially for PD and for shallower depths, indicating that our application of the pedotransfer function was successful in terms of reasonably estimating specific yield. The graph for PW shows a decreasing trend in the PPT/RSP ratio for the limited range it covers, which is in agreement with the specific yield graph (Ahmad, Hörmann et al., 2020; Chin, 2008; Heliotis, 1989; Van Gaalen et al., 2013; Zhang et al., 2017). The PPT/RSP ratio over depth shows a good agreement with specific yield, as derived from the bulk density data, especially for PD and for shallower depths, indicating that our application of the pedotransfer function was successful in terms of reasonably estimating specific yield. The graph for PW shows a decreasing trend in the PPT/RSP ratio for the limited range it covers, which is in agreement with the specific yield graph.

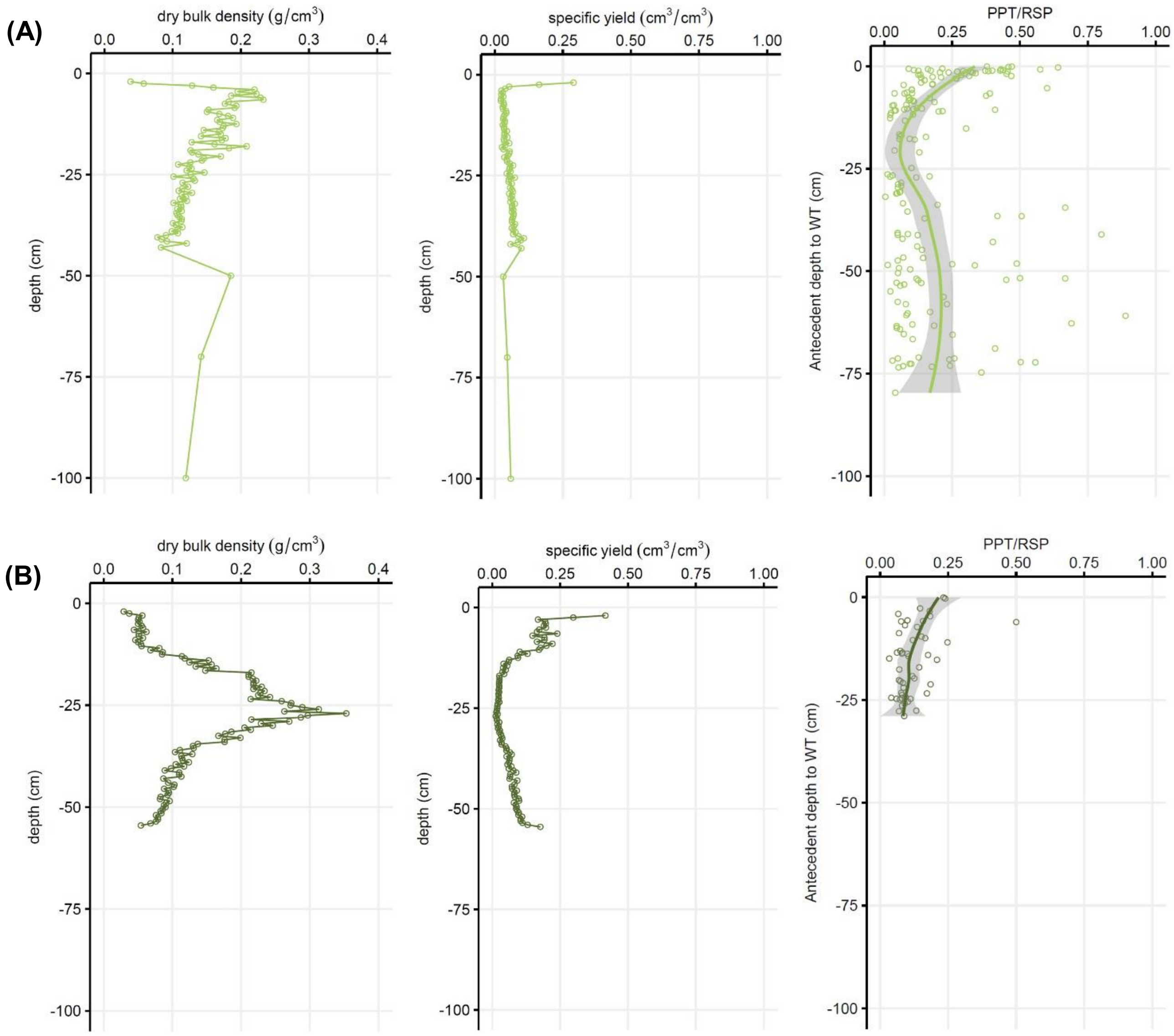


Figure 2.6 Depth profiles of peat dry bulk density, specific yield (determined from bulk density), and the ratio of precipitation event size to water table response height (PPT/RSP) for (A) PD (B) PW. PPT/RSP was smoothed using locally estimated scatterplot smoothing (loess) function, with the shaded areas showing confidence intervals.

Table 2.4 Average estimated specific yield per depth class.

SITE	PD		PW	
Depth (cm)	**Mean**	**SD**	**Mean**	**SD**
0–10	0.06	0.07	0.21	0.07
10–20	0.04	0.01	0.07	0.05
20–30	0.06	0.02	0.02	0.00
30–40	0.07	0.00	0.05	0.02
40+	0.08	0.04	0.09	0.03

2.4. Discussion

Our study found evidence that rewetting of a previously drained and degraded percolation fen may alter the physical response of the groundwater table to precipitation, increasing the hydrological buffer function of the peatland. Similar to our finding, the rise in water table in reaction to precipitation during storm events, was larger in a drained than in a rewetted blanket bog site (Holden et al., 2011). Another study conducted in a blanket bog catchment showed that drainage of peat resulted in a greater sensitivity to rainfall with flashier hydrographs than control catchments (Holden et al., 2006).

The slower response rate of the groundwater table to precipitation at the rewetted fen (PW) compared to the drained fen (PD) may be attributed to a difference in physical properties such as bulk density and specific yield. The long-term drainage at PD caused oxidation of organic matter and shrinkage of peat above the water table and compression below it (Kennedy & Price, 2004; Oleszczuk & Brandyk, 2008), which led to a decrease in the void ratio, and an increase in bulk density and thereby a shift in pore size distribution. In the absence of water-saturated conditions, aerobic decomposition and humification resulted in the degradation of soil structure and a decrease in water storage and water transmission (Kechavarzi et al., 2010). These processes led to the degraded peat in the upper horizons to have higher bulk density and a lower specific yield, compared to the peat in PW. At PW, soil properties changed because (1) the peat swelled up after rewetting and (2) new porous material ("proto-peat") accumulated over the 20 years since rewetting (Mrotzek et al., 2020).

The water storage capacity of peat is determined by its specific yield (Heliotis, 1989). At PW, the newly accumulated peat in the upper horizons has a much higher specific yield and therefore can store much more water than at PD. As a result, it requires a larger precipitation input to raise the groundwater level by the same height at PW, compared to PD. A positive feedback loop may establish in which the large storage capacity of the newly accumulated material buffers the water table response, creating conditions that are favorable for continued accumulation of poorly decomposed, coarse materials that help to maintain a large storage capacity (see Joosten & Clarke, 2002, p. 28).

Michaelis et al. (2020), who studied macroscopic and microscopic remains in the same peat cores from PD and PW, concluded that rewetting in PW had led to the establishment of an accumulating ecosystem ("proto-peat" formation). A mass-balance study on the same two study sites by Mrotzek et al. (2020) found that 11 cm of peat corresponding to 4.5 kg m^{-2} of organic matter, had accumulated over the 20 years since rewetting. As expected, the drained site PD had a much lower mean water table position than the rewetted site PW (−367 mm vs. +39 mm from the peat surface, respectively).

The drained fen (PD) has a less predictable groundwater table response to precipitation than the rewetted fen (PW), as indicated by the lower R-squared value of the regression model for PD. In other words, while in PW, precipitation intensity can explain more than 90% of the variation in groundwater table response rate, in PD, it alone explains only 71%. Particularly in PD, factors other than just rainfall likely play a distinct role as well, in determining water table dynamics. Among these factors is a differentiated specific yield over depth.

The pedotransfer function to convert peat bulk density to specific yield worked reasonably well. It is consistent with the PPT/RSP values, which has been shown to be a good indicator of specific yield (Ahmad, Hörmann et al., 2020; Heliotis, 1989; Holden et al., 2011; Van Gaalen et al., 2013). Any disagreement between the estimated specific yield values and the values of PPT/RSP for the same depths can be explained by variable initial conditions at the onset of a precipitation event. Heliotis (1989) in a study of forested peatland states that such variable conditions include, among other factors, antecedent soil moisture content and changes in biomass - affecting evapotranspiration and rainfall interception - through the growing season. As such, PW with taller vegetation may have marginally higher evapotranspiration and rainfall interception than PD, where vegetation is shorter. However, we do not expect vegetation at the study sites to play a major role in modifying the effect of precipitation on groundwater response.

The general tendency of decreasing specific yield with depth is in agreement with other studies (Ahmad, Hörmann et al., 2020; Bourgault et al., 2017; Heliotis, 1989; Menberu et al., 2016). While our specific yield estimates range from about 0.01 to 0.41 for the rewetted site (PW), Heliotis (1989) estimated a specific yield within a range of 0.05 to 0.7. Similarly, Menberu et al. (2016) found a higher mean specific yield of 0.14 within 20 cm of the peat surface, which decreased to 0.09 beyond 20 cm depth.

While our method of using hydrological monitoring data to analyze groundwater response to precipitation provides a fast and simple way to assess restoration success of fens, at least in terms of the physical properties in the upper horizons of the peat body, several limitations exist. First, we assumed any effect of evapotranspiration on the groundwater response to be negligible within the temporal scale of precipitation events. Second, as we had only two monitoring wells at only two sites, care must be taken in drawing generalized conclusions, particularly concerning other sites with different drainage histories or land-use. Although more monitoring wells could have benefitted the study, water levels measured in an observation well are representative of an area of at least several tens of square meters (Maréchal et al., 2006). A "before-after-control" approach, such as the one used by Menberu et al. (2016) on boreal peatland restoration, would have allowed a true assessment of the hydrological buffer functions at the restored site. However, assessing the effects of restoration over longer time frames (decades) would require long-term water level monitoring, which is generally lacking for (temperate) fens (cf. Bechtold et al., 2014).

2.5. Conclusions

The rate of groundwater response to precipitation event intensity is more than two times slower in the rewetted than in the drained fen. This finding is consistent with our hypothesis. We conclude that long-term rewetting of drained and degraded peatlands has the potential to alter the physical response of groundwater table to precipitation events and improve hydrological buffering. We attribute such a change to the restoration of physical properties of the peat body, especially at the upper soil horizon, which increases the water storage capacity of the peat (as evidenced by the estimated specific yield), and decreases groundwater response. The mechanisms by which macroporosity and water storage are restored are either by new peat accumulation (due to favorable conditions), or by swelling caused by water saturation, or both. Other studies confirm that new organic material ("proto-peat") has been accumulating in the same fen since rewetting. Our approach provides a fast and simple way to utilize hydrological monitoring data to understand physical properties of peat, which is especially useful in the absence of soil data. Future research should make use of long-term hydrological data in a similar way to analyze restoration success, especially when the data can be divided into pre-rewetting and post-rewetting periods.

Author Contributions

Sate Ahmad: Conceptualization, Methodology, Formal analysis, Writing - original draft, Writing - review & editing, Visualization, Software, Investigation. **Haojie Liu:** Conceptualization, Supervision, Methodology, Formal analysis, Investigation, Data curation, Writing - review & editing, Visualization, Project administration. **Anke Günther:** Data curation, Software, Writing - review & editing, Investigation. **John Couwenberg:** Formal analysis, Investigation, Data curation, Writing - review & editing. **Bernd Lennartz:** Conceptualization, Supervision, Writing - review & editing, Funding acquisition, Project administration.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgments

We would like to thank Kim Perkins for her initial support with exploring our methodological options. We would also like to express our gratitude to Almut Mrotzek for sharing soil profile data on bulk density, Shajratul Alam for advice on statistical analysis, Sarah Schwieger, for providing insights into the vegetation of the two sites, Florian Beyer for providing advice on the map of the study sites, Christian Schmidt and Laurenz Teuber for hydrological and meteorological data transfer/collection. Finally, we would like to thank the editor and the four anonymous reviewers. The European Social Fund (ESF) and the Ministry of Education, Science and Culture of Mecklenburg-Western Pomerania funded this work within the project WETSCAPES (ESF/14-BM-A55-0028/16, ESF/14-BM-A55-0030/16 and ESF/14-BM-A55-0035/16). Intellectual support was also received from the Research Training Group "Baltic TRANSCOAST" funded by the Deutsche Forschungsgemeinschaft (DFG, German Research Foundation) under Grant Number DFG-GRK 2000.

Supplementary Material to Chapter 2

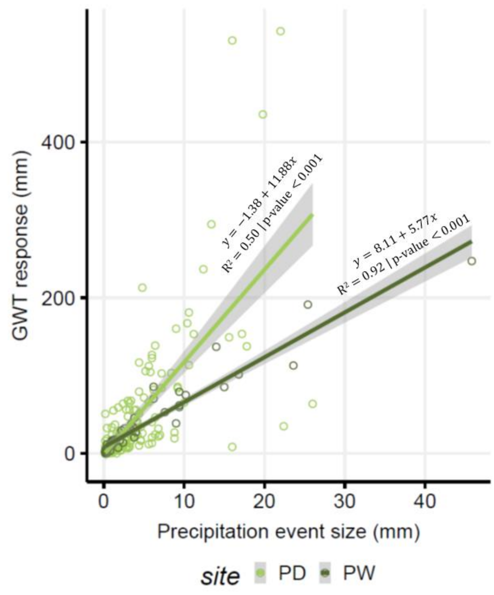


Figure SM 2.1 Groundwater table (GWT) response and precipitation event size in drained (PD) and rewetted (PW) fen. Shaded areas represent the 95 % confidence intervals.

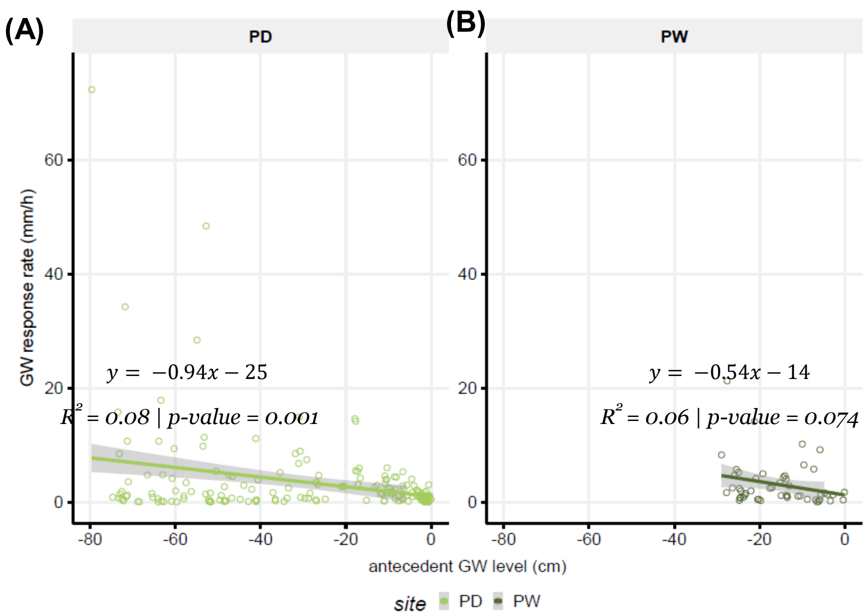


Figure SM 2.2 Groundwater (GW) response rate and antecedent groundwater level in **(A)** drained (PD) and **(B)** rewetted (PW) fen. Shaded areas represent the 95 % confidence intervals.

3. Meteorological controls on water table dynamics in fen peatlands depend on management regimes

Sate Ahmad[1*], Haojie Liu[1], Shajratul Alam[2], Anke Günther[3], Gerald Jurasinski[3], Bernd Lennartz[1]

[1]Soil Physics, Faculty of Agricultural and Environmental Sciences, University of Rostock, Rostock, Germany
[2]University of California, Santa Cruz, CA, USA
[3] Landscape Ecology and Site Evaluation, Faculty of Agricultural and Environmental Sciences, University of Rostock, Rostock, Germany

* Corresponding Author (~80 % contribution)

Abstract

Fens belong to the most threatened ecosystems in Europe. Maintaining a high water table through rewetting is an effective measure to rehabilitate many of their ecosystem functions. However, the impact of meteorological conditions such as relative humidity, precipitation, and air temperature on water tables is still unclear for rewetted fens. Here, we quantify the impact of meteorological factors on water table dynamics in a drained and a rewetted fen, using multiple time-series regression with data from continuous high-resolution (temporal) water level monitoring and weather stations. We find that a 1-degree rise in daily maximum air temperature causes a drop of about 4 mm in the water table in the drained and degraded fen but only a drop of around 2 mm at the rewetted fen, principally through evapotranspiration. Furthermore, higher minimum relative humidity limits evapotranspiration and thus causes a rise in the water table at both sites. Precipitation contributes to recharge, causing the water table to rise almost six times higher at the drained site than at the rewetted site. We attribute the differential influence of meteorological conditions on water table dynamics to (1) differences in vegetation which act as surface controls and (2) differences in soil properties. Our study underlines the importance of meteorological factors and surface controls for peatland restoration. Continuous monitoring of water-table and vegetation development in rewetted fen peatlands is advisable to ensure long-term success especially under climate change conditions and associated drought and dry spell events.

Keywords: peatland hydrology, peatland restoration, evapotranspiration, peatland hydrometeorology, diurnal groundwater fluctuation, climate change

3.1. Introduction

Over the past century, about 90% of minerotrophic peatlands (fens; groundwater-fed peatlands) in Central Europe have been degraded through artificial drainage and deforestation (Joosten & Couwenberg, 2001). As a result, fens have been recognized as one of the most threatened ecosystems in Europe (Schrautzer et al., 2007). Maintenance of water table at or near the peat surface prevents carbon mineralization, allows peat accumulation (Michaelis et al., 2020; Mrotzek et al., 2020), and improves ecosystem functions such as hydrological buffering, water purification, erosion protection and climate regulation (Ahmad, Liu, Günther et al., 2020; Günther et al., 2020; Kimmel & Mander, 2010; Lennartz & Liu, 2019). Therefore, restoration of degraded peatlands is an important climate change mitigation measure, as peatlands store much of the global terrestrial carbon stock. Although hydrological restoration of peatlands has been implemented throughout Europe and North America (Lamers et al., 2015), rewetted peatlands are under pressure from climate change (Levison et al., 2014). Shifting in precipitation patterns and increasing evapotranspiration resulting from global warming may further degrade peatlands (Helbig, Waddington, Alekseychik, Amiro, Aurela, Barr, Black, Blanken et al., 2020; Nijp et al., 2015; Tarnocai, 2009). However, the impact of a changing climate on peatland ecohydrology through extreme weather events such as droughts, heatwaves and dry spells is likely not uniform over different spatial scales and climatic zones. Part of this variability is due to a variation in local meteorological conditions and differences in land management.

Meteorological factors may affect the water-table in peatlands through several processes. Heavy precipitation can serve as a hydrological input in the peatland water balance through recharge (Cooper et al., 2019; Ferone & Devito, 2004; Menberu et al., 2016), while high temperatures can increase evapotranspiration (Bridgham et al., 1999). Humidity can also modify the effect of temperature on evapotranspiration because it is easier for water to evaporate into drier air than into more saturated air (Ahmad, Hörmann et al., 2020; Chattopadhyay et al., 2009; Shaw et al., 2010). The effect of meteorological parameters on water table fluctuation may also be modified by microtopography, vegetation, soil properties and land management (Baldocchi et al., 2004; Dunne et al., 1991). Peat soils are highly heterogeneous porous media (Ahmad, Liu, Beyer et al., 2020; Liu, Price et al., 2020; Rezanezhad et al., 2016) with hydraulic conductivities that may vary over two orders of magnitudes within the same peat horizon (Liu & Lennartz, 2019a). Peat pore structures are highly diverse (McCarter et al., 2020) while peat properties often exhibit anisotropy (Wang et al., 2020). The spatial differences in soil properties can cause different responses of the water table to precipitation at different locations within a peatland. Furthermore, peatland rewetting can alter the prevailing vegetation structure and composition (Malhotra et al., 2016; Schrautzer et al., 2013), which can further modify the interactions between meteorological factors and water table. For example, the relationship between temperature and water loss may be modulated by stands of dominant vegetation, with high evapotranspirative demand (Bridgham et al., 1999).

The majority of published studies on the link between peatland ecohydrological processes and meteorological conditions so far focus on bogs (ombrotrophic peatlands or rainfed mires; Bourgault et al., 2019; Philippov & Yurchenko, 2019; Price, 1996; Ruseckas & Grigaliūnas, 2008), while similar studies on fens are sparse. Therefore, meteorological effects on water table dynamics in fens, especially with a focus on different management measures (such as rewetting or artificial drainage), are understudied. To address these shortcomings, we (1) investigate how on-site meteorological conditions act as controls over water-table dynamics (2) unravel the underlying mechanisms involved and (3) evaluate how these hydrological controls differ over different management regimes. To this end, we characterize water table dynamics, estimate the effect of daily rainfall, relative humidity and temperature on the water-table and quantify rain-free day actual evapotranspiration at a drained fen and a rewetted fen in North-east Germany.

3.2. Methods

3.2.1. Study Sites

The two study sites (drained fen, PD and rewetted fen, PW) are located in the federal state of Mecklenburg-Vorpommern, Germany (**Figure 3.1**). They are 8 km apart and together belong to one of the largest connected fen complexes in northeastern Germany (Jurasinski et al., 2020). According to the hydrogenetic mire classification system (Joosten et al., 2017; Succow & Joosten, 2001), they are 'percolation fens' which are minerotrophic peatlands that depend on a large supply of water that is distributed evenly throughout the year. Percolation fens are often located along river valleys which are remains of meltwater channels of the last glaciation, where permanent groundwater flow from adjacent moraines caused paludification (Koch & Jurasinski, 2015). Both sites were drained before 1750. In PW, land drainage was intensified around 1970 for high-intensity pasture management. In 1997, the site PW was rewetted as a part of the state peatland conservation program and the EU-LIFE program, while PD remains drained.

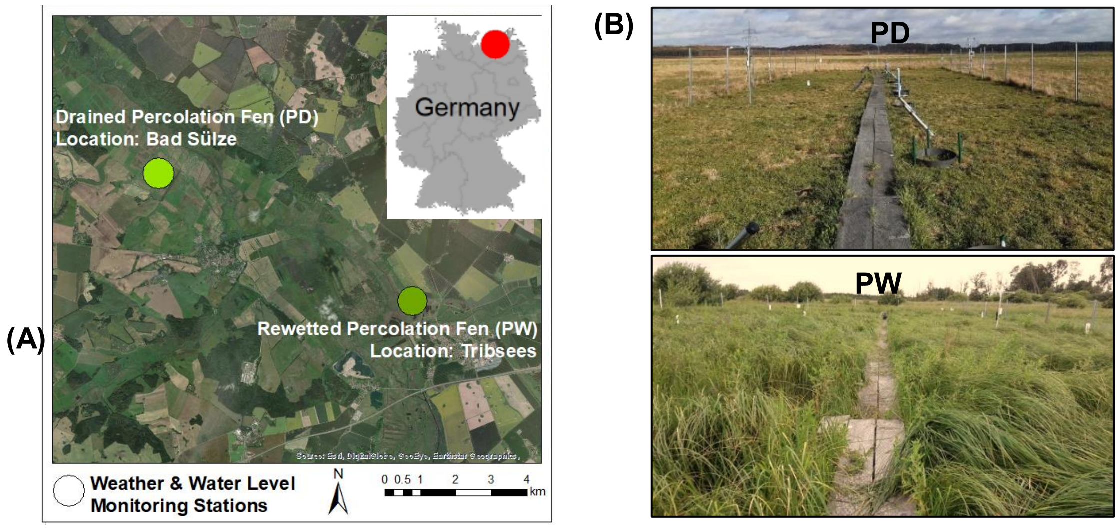


Figure 3.1 (A) The study sites include a drained fen and a rewetted fen in Mecklenburg-Vorpommern. (B) The study plots at PD (upper panel) and PW (lower panel). The weather station at PD appears on the left of the photo, while at PW it appears on the right. Photo: Haojie Liu (PD) and Michael Franz (PW).

PD can be considered to be a fairly homogenous grassland, with a dominance of *Ranunculus repens L.* and *Deschampsia cespitosa* (L.) P. Beauv. with some *Holcus lanatus* L. and *Poa trivialis L.* PW is much more diverse with a mosaic of several dominant stands that established after rewetting. The studied plot in PW is dominated by *Carex acutiformis* Ehrh., with few occurrences of *Epilobium hirsutum* L. In recent times, PW can be considered to feature near-natural percolation fen vegetation and is part of a larger valley mire system (Tiemeyer et al., 2006). The relevant site characteristics have been described by Jurasinski et al. (2020) and by Ahmad, Liu, Günther et al. (2020).

3.2.2. Data acquisition and analyses

As part of the WETSCAPES project, described by Jurasinski et al. (2020), one water level logger (SEBA hydrometrie Dipper-PT) was installed for each site and data was registered every 15 min. Weather stations were also installed at each site, which recorded precipitation (Rain Collector #07852, Davis Instruments), relative humidity and temperature (KPK 1/5-ME, Galltec Mess- und Regeltechnik GmbH and MELA Sensortechnik GmbH) every 10 min using an automated data logger CR300 from Campbell Scientific.

All data were merged and temporal scales were matched to 30-minute intervals. All data analyses were carried out using the R version 4.0.3 (R Core Team, 2019). For our analysis, we used data from 22 September 2017 through 19 August 2020 (2.9 years) for both sites. The dataset was aggregated to daily resolution by calculating daily precipitation sums and averaging all other variables. Multiple linear regressions were carried out using the "lm" function in R, by setting water table as the dependent variable, and minimum relative humidity, maximum air temperature and total precipitation as the independent variables. Seasonality was statistically adjusted for by using dummy/indicator variables of each month (11 months in total) with January as the reference month. This approach is commonly used to control for seasonality or periodicity (Hyndman et al., 2020 - Chapter 5.4; Maki et al., 1978; Hylleberg et al., 1993). The summary statistics of the key variables are provided in **Table 3.1.**

Table 3.1 Mean values of daily water level and meteorological variables for PD and PW. The square brackets contain the lower and upper confidence intervals.

Variables	PD	PW
Mean Daily Water Level (mm)	-277.96 [-292.75, -263.18]	2.88 [-3.71, 9.46]
Maximum Daily Air Temperature (°C)	13.70 [13.19, 14.20]	14.59 [14.09, 15.09]
Minimum Daily Relative Humidity (%)	65.47 [64.31, 66.62]	64.10 [62.92, 65.27]
Daily precipitation (mm)	1.28 [1.09, 1.47]	1.54 [1.31, 1.76]

3.2.3. Evapotranspiration and specific yield determination

To further investigate the underlying processes of how temperature and humidity impact water-table dynamics, rain-free day evapotranspiration (ET) from May to October was estimated for both sites using the Hays method (Hays, 2003), which is a modification of the White method (White, 1932). For ecosystems with shallow water tables, groundwater use by vegetation through evapotranspiration (ET) can be estimated by analyzing water table fluctuation which is relatively simple to implement (Ahmad, Hörmann et al., 2020; Mazur et al., 2014; Mould et al., 2010). Water level fluctuation methods can be useful to apply in wetland ecosystems for estimating ET especially because it integrates several factors, including the growth-cycle of plants, the plant types, and moisture availability which are generally missing from micrometeorological methods of ET determination (Lautz, 2008; Mazur et al., 2014). The equation for ET determination is as follows:

$$\boldsymbol{ET} = \left[(\boldsymbol{H_1} - \boldsymbol{L}) + \frac{(\boldsymbol{H_2} - \boldsymbol{L})}{\boldsymbol{T_1}} \times \boldsymbol{T_2}\right] \times \boldsymbol{S_y} \qquad \textbf{(3.1)}$$

where, ET = evapotranspiration rate (mm/day); H_1 = highest groundwater level (mm) on an observed day (usually early morning), H_2 = highest groundwater level on the day after the observed day (mm), L = lowest groundwater level (mm) on an observed day (usually evening), T_1 = time between L and H_2 (rising period), T_2 = time between H_1 and L (drawdown period), S_y = mean specific yield (dimensionless) for the observed diurnal groundwater fluctuation (see an example of diurnal groundwater fluctuation in **Figure 3.2** with respective equation parameters being labelled accordingly).

The daily water uptake by plants is calculated as the water level difference between the low *L* and the high *H1* of the given day. This period of groundwater level fall (T2) is the 'drawdown period' which includes water uptake by plants and net groundwater inflow. The second component of the equation is then added, which quantifies the latent water rise caused by inflow during the drawdown period. A key assumption is that ET is zero during this time (Hays, 2003; Mould et al., 2010). Evapotranspiration was estimated using the dataset with a 30-minute resolution. A user-defined function was developed in R to determine evapotranspiration according to **equation (3.1)**, which calculates H1 as the maximum water level of a given day, L as the minimum water level of the given day and H2 as the maximum water level of the following day. If the values of T1 and T2 did not add up to 24 hours for a given diurnal groundwater fluctuation, the values of ET were corrected to represent 24 hours. The function was applied only to days with diurnal fluctuation in the absence of rainfall in both study sites, evaluated using a graphical method. In total, daily ET could be determined for the same 55 days for PD and PW. The Shapiro-Wilk test showed that the differences of ET between the PD and PW are normally distributed (W=0.96; p-value=0.108). Thus, statistical testing was done using paired t-test, pairing the ET values of the sites by the same day.

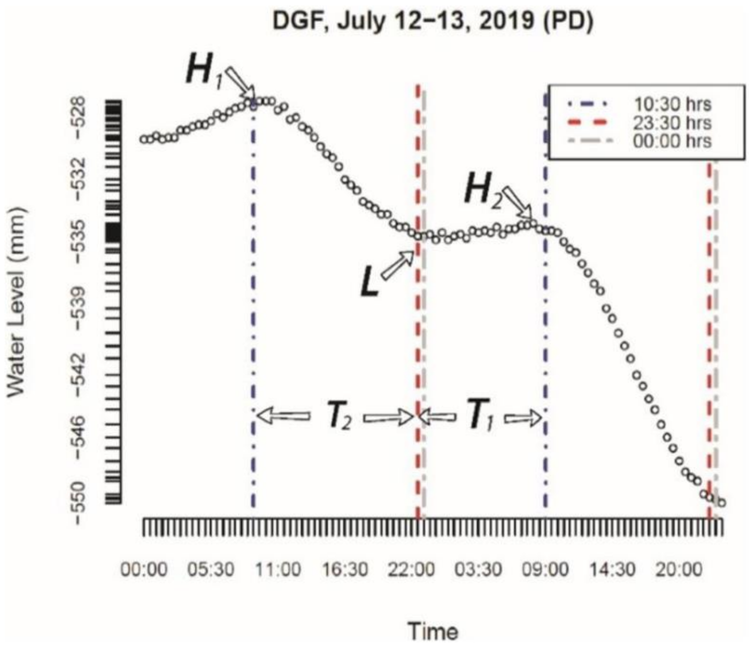


Figure 3.2 An example of diurnal groundwater fluctuation (DGF) on a rain-free day at PD.

Specific yield, which is the amount of water that would drain from a unit of soil if the water table would drop by a unit height (Childs, 1969), is a required parameter to determine ET when using the diurnal groundwater fluctuation method (see above). To determine specific yield we used the following equation according to Dolan et al. (1984) and Ahmad, Hörmann et al. (2020):

$$S_y = \frac{P_{re}}{\Delta GWL} \qquad (3.2)$$

where, S_y = specific yield, P_{re} = quantity of precipitation in mm, during the rainfall event, ΔGWL = change in groundwater level which is the difference between the water level at the start of the precipitation event and the water level at the end of the event.

P_{re} and ΔGWL were calculated using a user-defined function following Ahmad, Hörmann et al. (2020) and Ahmad, Liu, Günther et al. (2020). The function estimates the quantity of precipitation during precipitation events (in mm) and the respective event duration (in h) using a temporal moving window of 6 h. Summation of precipitation quantity and event duration stopped if precipitation ceased for 30 minutes. The initial and final water levels were recorded, and the difference (ΔGWL) was calculated for each event. The specific yield was plotted against the mean water level (determined as the mean of the initial and the final water level for each precipitation event, **Figure 3.3**). All events with water levels above the peat surface and all negative values of ΔGWL (indicating declining water levels despite ongoing precipitation) were removed. Values which did not fall between 0 and 1 were also removed as they violate the definition of specific yield. The effect of antecedent soil moisture was assumed to be negligible. A certain rainfall intensity is required for initiating a water table response. We filtered PD to only include rainfall events of at least 0.33 mm h^{-1} intensity. We reduced the threshold for PW to be able to include more events, since PW, being a rewetted site, has water level at the surface for most of the year. Thus, events with intensities of at least 0.1 mm h^{-1} were included for PW. The final number of rainfall events for PD is 73 and for PW is 65. Lowess smoothing function (“loess”, R Core Team, 2019) was used to determine the specific yield for a given mean water level which is indicative of peat depth. The mean specific yield for a given diurnal fluctuation was plugged into the ET **equation (3.1)** depending on the mean water level for the diurnal fluctuation.

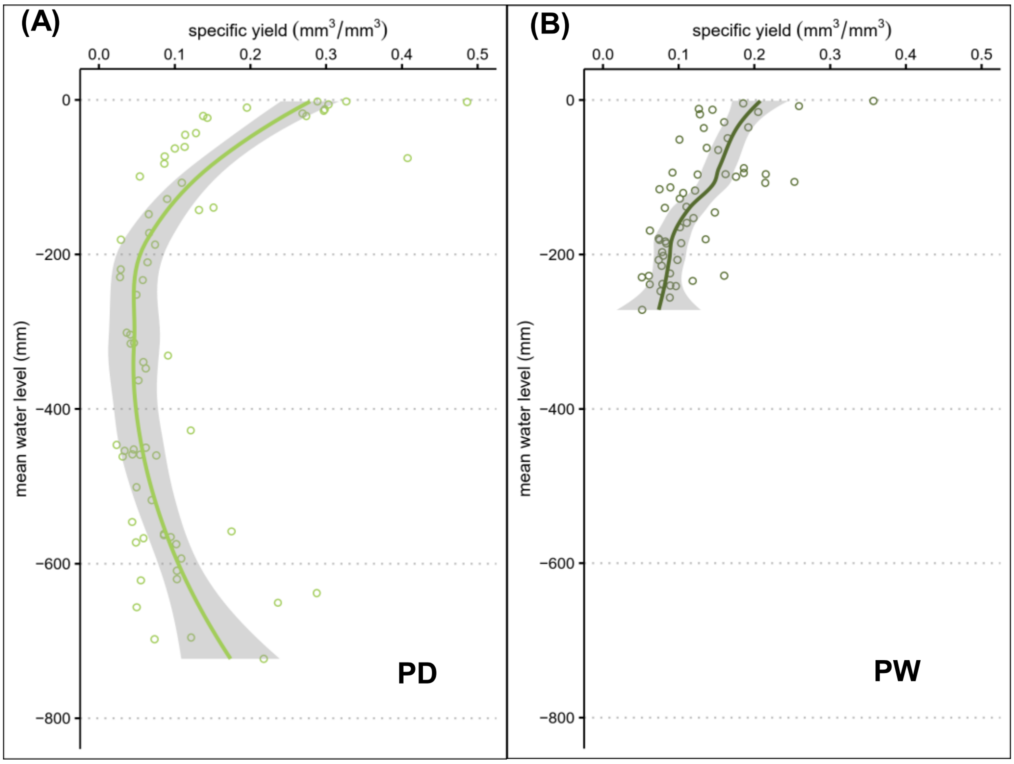


Figure 3.3 Specific yield determined as the precipitation event size (mm) to groundwater table response ratio (P_{re}: ΔGWL) according to mean water level for **(A)** PD and **(B)** PW. The solid lines represent the loess smoothing function and the shaded area represents the 95% confidence interval.

3.3. Results

3.2.4. Water table characteristics and meteorological conditions

Both sites show a clear seasonality in water tables as well as in meteorological factors (Figure 5 and 6). The water level at the drained fen (PD), is below the peat surface for most of the year, with a substantial drop from the middle of spring to the end of summer for all 3 observed years. Even at the rewetted fen (PW), the water table receded substantially to about -300 mm in the summers of 2018 and 2019. Receding water tables at both sites occur at times of rising daily mean air temperatures and an absence of substantial rainfall **(Figure 3.4** and **Figure** 3.5). The exceptionally dry late autumn and early winter seasons of 2018 and 2019, caused water levels to stay well below the peat surface at PD. For PD, the maximum daily rainfall (43 mm) occurred in October 2017 and for PD in July 2018 (46 mm). Autumn and winter seasons are characterized by high daily minimum humidity.

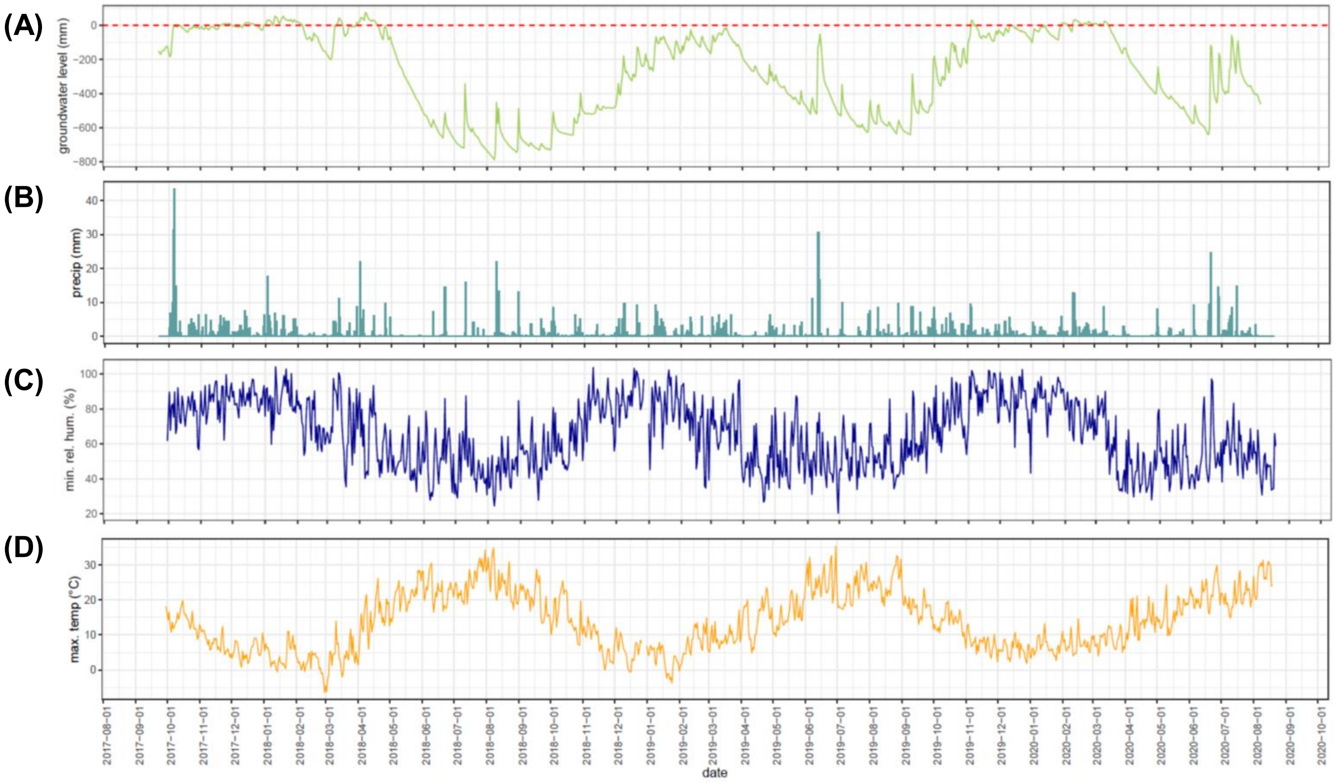


Figure 3.4 Daily time-series of **(A)** water level **(B)** precipitation **(C)** minimum relative humidity **(D)** mean air temperature at PD

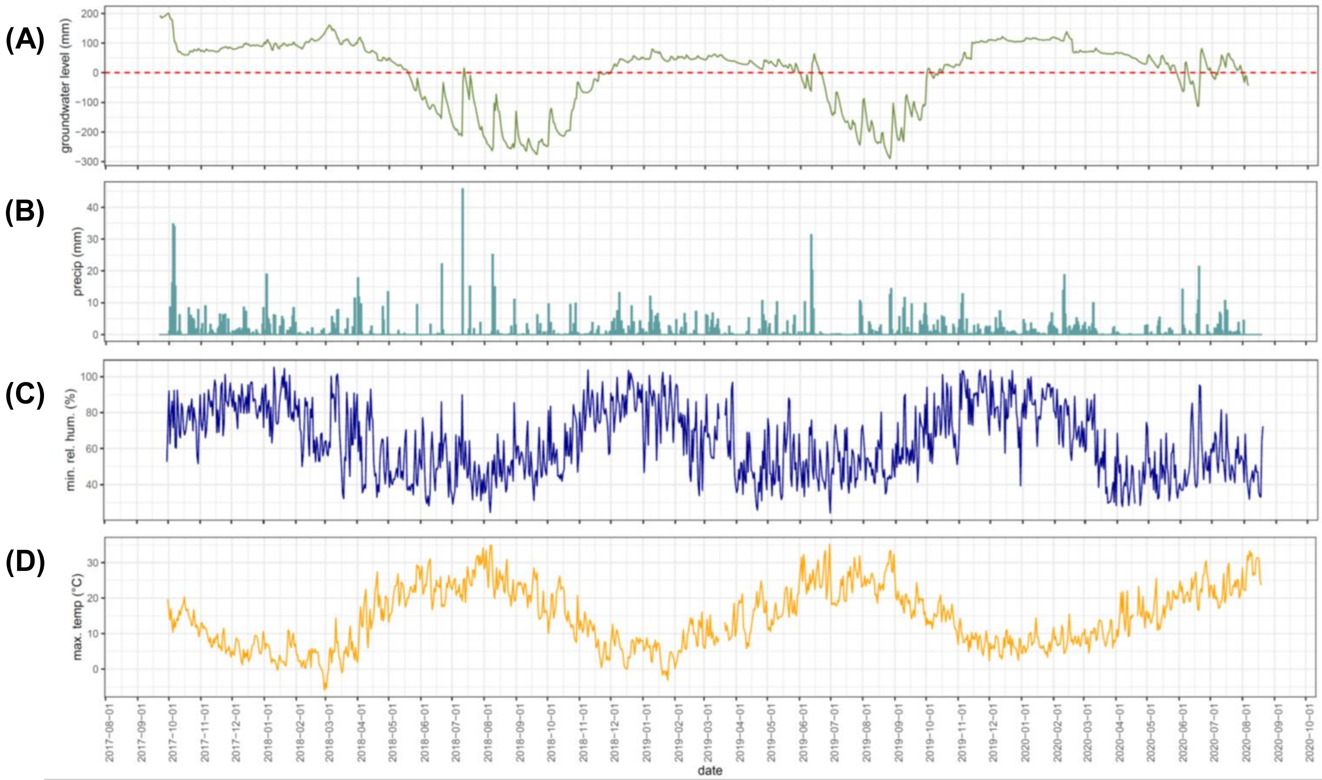


Figure 3.5 Daily time-series of **(A)** water level **(B)** precipitation **(C)** minimum relative humidity **(D)** mean air temperature at PW.

3.2.5. The effect of meteorological factors on water-table

The water table at both sites shows an immediate response to rainfall. At PD, for each degree Celsius increase in daily maximum temperature, the water level falls by about 4 mm. For each unit increase in daily minimum relative humidity, the water table rises by more than 2 mm. A unit rise in precipitation induces water level to rise by about 6 mm. All meteorological regression terms are significant. The model helps to significantly explain more than 66 percent of the variation in the water level at PD.

At PW, one unit rise in daily maximum air temperature causes the water level to drop by about 2 mm, which is almost half the drop in water level at PD. One unit increase in relative humidity only marginally increases the water level by around 0.8 mm. One millimeter increase in daily precipitation induces only a 1.4 mm increase in the water level, an effect which is substantially lower than at PD. This model explains around 60 percent of the variation in water level at PW. A graphical representation of the estimated coefficients is provided in **Supplementary Material (Figure SM 3.1).**

Table 3.2 Multiple linear regression of water level at PD and PW.

Sites	Regression Terms	Estimated Coefficients	Std. Error	t value	Pr(>|t|)	
PD	Intercept	-252.68	36.63	-6.87	1.09×10^{-11}	***
	January (reference)					
	February	37.89	21.79	1.74	0.082355	.
	March	48.57	21.80	2.23	0.026126	*
	April	-12.35	24.98	-0.49	0.621190	
	May	-188.81	25.82	-7.31	5.16×10^{-13}	***
	June	-277.77	29.24	-9.50	$< 2 \times 10^{-16}$	***
	July	-293.00	28.66	-10.22	$< 2 \times 10^{-16}$	***
	August	-372.92	31.11	-11.99	$< 2 \times 10^{-16}$	***
	September	-377.70	26.45	-14.28	$< 2 \times 10^{-16}$	***
	October	-167.10	23.38	-7.15	1.65×10^{-12}	***
	November	-120.35	21.33	-5.64	2.16×10^{-8}	***
	December	-50.99	20.89	-2.44	0.014811	*
	Daily Max. Air Temp. (°C)	**-3.89**	**1.10**	**-3.55**	**0.000397**	***
	Daily Min. Rel. Humidity (%)	**2.46**	**0.38**	**6.51**	$\mathbf{1.13 \times 10^{-10}}$	***
	Daily Precipitation (mm)	**6.15**	**1.44**	**4.26**	$\mathbf{2.26 \times 10^{-5}}$	***

Residual standard error: 142.3 on 1048 degrees of freedom | Multiple R-squared: 0.669 | Adjusted R-squared: 0.665 | F-statistic: 151.2 on 14 and 1048 DF | p-value: $< 2.2 \times 10^{-16}$

Sites	Regression Terms	Estimated Coefficients	Std. Error	t value	Pr(>|t|)	
PW	Intercept	27.00	17.68	1.53	0.126963	
	January (reference)					
	February	12.49	10.73	1.16	0.244983	
	March	15.44	10.79	1.43	0.152687	
	April	14.58	12.46	1.17	0.242180	
	May	-11.07	13.00	-0.85	0.394619	
	June	-67.88	14.82	-4.58	$5.17x10^{-6}$	***
	July	-108.52	14.59	-7.44	$2.12x10^{-13}$	***
	August	-166.94	15.78	-10.58	$< 2x10^{-16}$	***
	September	-175.80	13.83	-12.71	$< 2x10^{-16}$	***
	October	-72.98	11.68	-6.25	$5.94x10^{-10}$	***
	November	-36.59	10.45	-3.49	0.000513	***
	December	-6.27	10.27	-0.61	0.541946	
	Daily Max. Air Temp. (°C)	**-2.20**	**0.59**	**-3.75**	**0.000184**	***
	Daily Min. Rel. Humidity (%)	**0.82**	**0.18**	**4.58**	**$5.26x10^{-6}$**	***
	Daily Precipitation (mm)	**1.40**	**0.62**	**2.27**	**0.023242**	*

Residual standard error: 69.85 on 1048 degrees of freedom | Multiple R-squared: 0.593 | Adjusted R-squared: 0.598 | F-statistic: 111.4 on 14 and 1048 DF | p-value: < $2.2x10^{-16}$
*Signif. codes: <0.001***, <0.01** , <0.05*, <0.1 ·*

3.2.6. Actual evapotranspiration and specific yield during rain-free days

Daily evapotranspiration (ET) for 55 rain-free days was estimated for PD and PW. Mean ET at PD was 1.96 mm d^{-1} (minimum-maximum: 0.70 mm d^{-1} - 4.11 mm d^{-1}), while at PW mean ET was much higher (4.79 mm d^{-1}, minimum-maximum: 2.16 mm d^{-1} - 8.73 mm d^{-1}). Paired t-test reveals that ET at PW was significantly higher than at PD for the same days (mean difference = 2.82 mm d^{-1} [95% CI= 2.29 and 3.34], **Figure 3.6A**). Evapotranspiration at both sites differed over the months, with PW showing higher evapotranspiration compared to PD, for any given month (**Figure 3.6B**).

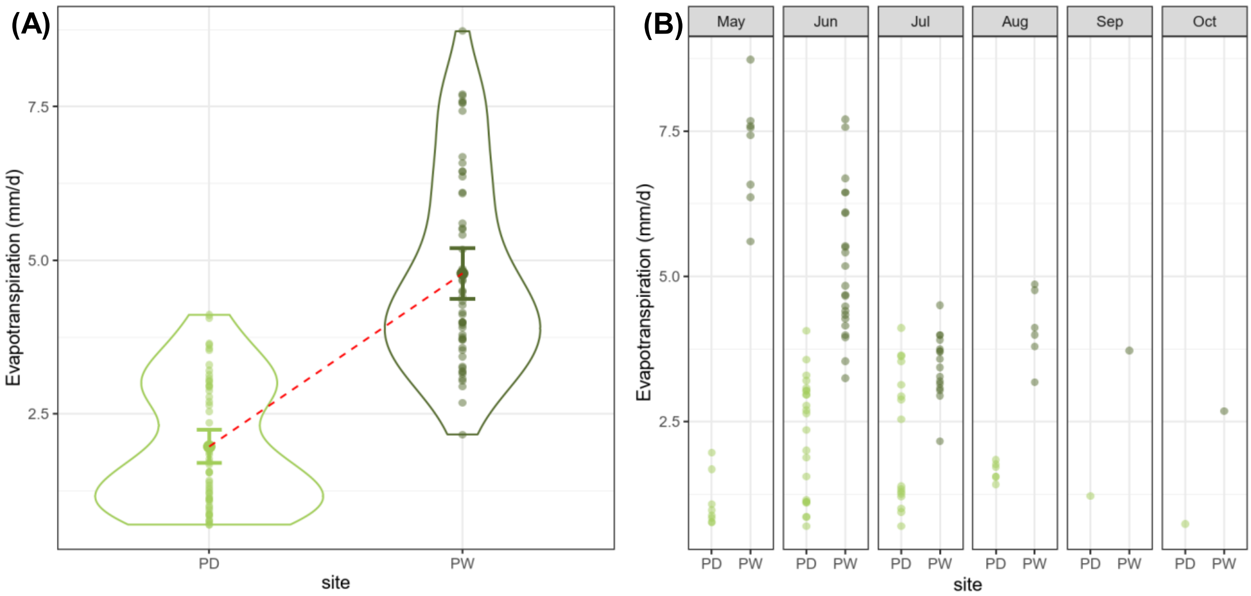


Figure 3.6 (A) Violin plots of ET with mean and error bars (95% confidence interval). The red dotted line connects the two means (of PD and PW). The violin shapes show the distribution of the data for each site. **(B)** Evapotranspiration for both sites according to months (in panels).

The average (median) specific yield values with which ET values were estimated were significantly higher at PW (0.284) than at PD (0.229, Wilcoxon rank-sum test with continuity correction, W= 1074, p-value = 0.008). Furthermore, the average (median) fluctuation (calculated as the difference between H1 and H2) for these days was significantly higher at PD (10 mm) compared to PW (8.4 mm, Wilcoxon rank-sum test with continuity correction, W=1850, p-value= 0.044).

3.4. Discussion

According to our data, precipitation, air temperature and humidity seem to affect the water table (significantly) at different magnitudes, depending on the drainage status of the fen, while the direction of the relationship between the meteorological variables and water levels were according to expectations. This was the case even though the local meteorological conditions at both the drained and rewetted fens were comparable.

Precipitation had a positive effect on water levels at both sites. This is not astonishing, since it is the primary way of groundwater recharge (Wittenberg et al., 2019). After infiltration at the peat surface, the water percolates through micro- and macropores and subsequently raises the water level (Heliotis & DeWitt, 1987). However, the magnitude of water table response at PD is more almost six times higher than at PW. In our recent study (Ahmad, Liu, Günther et al., 2020), we investigated the rate of water table rise as a response to discrete rainfall events at the same sites and also found that the water table response at PD is much higher than at PW. We attribute these differential effects to a difference in the specific storage capacity of the peat.

Daily minimum relative humidity had a positive effect on the water level. Higher humidity means that air with high water vapor can only store a low quantity of additional water and as such evapotranspiration becomes limited (Shaw et al., 2010). In other words, it is easier for water to evaporate into drier air, than into wetter air (Chattopadhyay et al., 2009). Thus, in the absence of high rates of evapotranspiration, the water level fluctuation is less pronounced. The higher the air temperature, the more water vapor it can hold and thus more water can evaporate (Kirschbaum & McMillan, 2018; Shaw et al., 2010). Furthermore, stomatal conductance increases with rising temperatures (Urban et al., 2017) and as a result, transpiration rates increase (especially when there is no water stress). Water loss through evapotranspiration causes the water level to drop and as such we find a negative relation between air temperature and water levels.

Although the meteorological conditions are comparable at the two sites, evapotranspiration at PW was, on average, twice as high as at PD. Therefore temperature and relative humidity cannot explain the differential effects at the two sites. A likely explanation of the differential effects of meteorological factors can be derived from understanding the difference in soil properties. The specific yield (averaged over the days for which ET was calculated) of peat at PD was significantly lower than that of PW, while the diurnal fluctuation at PD was significantly higher than at PW (although the ET derived at PD was significantly lower than at PW). This means that one unit change in the water-level at PD corresponds to a much lower volume of water than what the same unit drop or rise corresponds to in PW. Therefore, the higher impact of temperature and humidity on the water table at PD could, in part, be due to the lower storage capacity of the degraded fen peat.

Another explanation, however, could be the difference in vegetation as it has been shown to influence evapotranspiration rates (Jimenez-Rodriguez et al., 2019). The vegetation at PD is much shorter (around 20 cm in height) and mostly comprised of grasses, while the vegetation at PW is dominated by taller sedges (*Carex acutiformis*), around 1 m in height (see Schwieger et al., 2020). Schwieger et al. (2020) quantified plant production in both these sites and report a much higher plant biomass production at PW (aboveground: 346 g m^{-2} y^{-1}; belowground: 199 g m^{-2} y^{-1}) than at PD (aboveground: 80 g m^{-2} y^{-1}; belowground: 43 g m^{-2} y^{-1}). Such substantially higher plant biomass not only helps to explain the much higher ET at PW but could also explain why PW shows less sensitivity to changes in relative humidity and air temperature. The taller vegetation and much higher biomass in PW likely create a below-canopy microclimate that is much different than the above canopy meteorological conditions. For example, a tall and dense canopy can provide substantial shade and cause a cooling effect at the soil surface (dissipation of incident solar radiation, see Huryna et al., 2014). Additionally, it can also decrease the escape of water vapor to the atmosphere, increasing humidity under the canopy. As such, values of relative humidity and air temperature registered by the weather station at the study areas (especially at PW) placed above the canopy may not accurately reflect the values closer to the soil surface (below-canopy). Therefore, the vegetation at PW likely acts as a surface control, thereby modulating the magnitude at which above canopy temperature and humidity affect water table dynamics. Such effect modification is likely to be lacking in PD as its vegetation is much shorter and biomass production much lower.

Other than differences in vegetation as a cause for differences in ET, the depth to water table can also provide additional explanation. During the study period, PD had a mean daily water level of around -30 cm while for PW the mean daily water level is nearly 0 cm (at the surface). Several studies show that high ET rates generally occur in sites with shallow water tables, and lower ET rates in sites with deep water tables (Cooper et al., 2006; Duell, 1988; Nichols, 1994; Zhang & Schilling, 2006).

While we were successful at quantifying the relationship between meteorological conditions and water table dynamics, several limitations need to be considered. As we had only two monitoring wells at only two sites, care must be taken in generalizing conclusions, particularly for fens with different drainage histories or with a different hydrological regime and/or vegetation. Although the study could have benefitted from a higher number of monitoring wells, water levels measured in an observation well are representative of an area of at least several tens of square meters (Maréchal et al., 2006). For a better understanding of the effect of rewetting or management regimes, a "before-after-control" approach could be more appropriate (see Menberu et al., 2016). However, such long-term monitoring of hydrometeorological parameters is especially rare for temperate fen peatlands (Ahmad, Liu, Günther et al., 2020; Bechtold et al., 2014). A final limitation is that our weather stations were located above the plant canopy, and thus were unable to capture below-canopy microclimatic differences. Future research projects involved in collecting data on meteorological parameters can benefit from the installation of microclimate sensors below the canopy that are now readily available at competitive costs (Lembrechts et al., 2020).

Our results underline the importance of meteorological effects on water table dynamics in fens and how such controls can be modified by the prevailing vegetation characteristics and soil properties, which in turn are influenced by management decisions. Our findings have implications for prompt and effective rewetting of drained and degraded peatlands. Degraded fen peatlands, because of their low water storage capacity are likely to be more vulnerable to temperature extremes which can cause water-tables to rapidly decline, thereby further amplifying the process of peat degradation and carbon mineralization. Therefore, delays in peatland rewetting are likely to result in additional efforts and resources being required to restore ecosystem functions. The longer we wait with rewetting, the harder it will be to achieve water levels continuously fluctuating around the peat surface. For rewetted fens, continuous monitoring is advisable to ensure long-term success especially under a changing climate and associated heatwaves and dry spell events.

Conflict of Interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Author Contributions

SA: Conceptualization, Methodology, Formal analysis and coding, Writing - original draft, Writing - review & editing, Visualization, Software, Investigation. **HL**: Conceptualization, Supervision, Methodology, Formal analysis, Investigation, Writing - review & editing, Visualization, Project administration. **ShA**: Formal analysis and coding, Writing- review & editing. **AG**: Data curation, Writing - review & editing, Investigation. **GJ**: Formal analysis and coding, Funding acquisition, Writing - review & editing. **BL**: Conceptualization, Supervision, Writing - review & editing, Funding acquisition, Project administration.

Funding

The European Social Fund (ESF) and the Ministry of Education, Science and Culture of Mecklenburg-Western Pomerania funded this work within the project WETSCAPES (ESF/14-BM-A55-0028/16, ESF/14-BM-A55-0030/16 and ESF/14-BM-A55-0035/16). Intellectual support was also received from the Research Training Group "Baltic TRANSCOAST" funded by the Deutsche Forschungsgemeinschaft (DFG, German Research Foundation) under Grant Number DFG-GRK 2000.

Acknowledgments

We would like to thank Daniel Köhn, Christian Schmidt and Laurenz Teuber for hydrological and meteorological data transfer/collection. We would also like to express our gratitude to Sarah Schwieger, for providing insights into the vegetation characteristics of the two sites. Finally, we would like to thank the editor and the anonymous reviewers.

Supplementary Material to Chapter 3

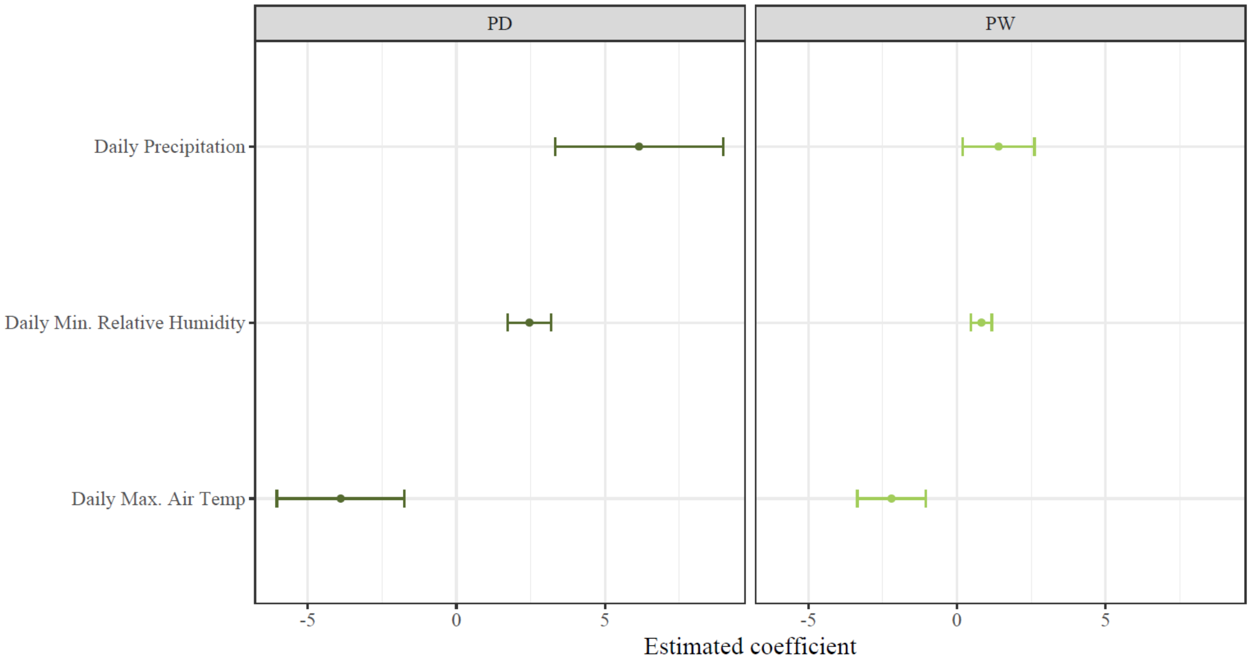


Figure SM 3.1 The differential effects of meteorological factors on the water level at PD and PW, as indicated by the estimated coefficients and their respective 95% confidence intervals (detailed regression results are provided in Table 2. A variable is significant if the mean and confidence interval does not overlap with zero. A negative estimated coefficient indicates a negative effect, while positive values indicate a positive effect on the water level.

4. Spatial heterogeneity of soil properties in relation to microtopography in a non-tidal rewetted coastal mire

Sate. Ahmad[1*], Haojie Liu[1], Florian Beyer[2] , Bjørn Kløve[3] , Bernd Lennartz[1]

[1]Soil Physics, Faculty of Agriculture and Environmental Sciences, University of Rostock, Germany
[2]Geodesy and Geoinformatics, Faculty of Agriculture and Environmental Sciences, University of Rostock, Germany
[3]Water, Energy and Environmental Engineering Research Unit, University of Oulu, Finland
*Corresponding Author (~80 % contribution)

Abstract

Over the past century, mires and peatlands have faced a wide range of degradation by artificial drainage, making them one of the most threatened ecosystems in Europe. However, restoration of drained peatlands has gained much importance over the last three decades, mostly due to the multiple ecosystem services they provide such as carbon storage, habitat provision and water flow regulation. Although there has been an increased focus on such ecosystems, spatial research on hydrophysical soil properties following rewetting in coastal mires is lacking. Therefore, the objectives of the study were to understand the spatial structures of hydrophysical properties of organic soils and spatial patterns of organic matter accumulation in relation to soil surface microtopography. Soil organic matter content (SOM) and hydraulic conductivity (K_s) of topsoils (0–28 cm), along with soil textures of the underlying mineral substrate, were investigated in a rewetted nontidal coastal flood mire (Baltic Sea). The results indicate that the organic horizon with its relatively low Ks acts as a hydrological barrier to infiltration. Soil organic matter content (SOM), Ks and soil surface microtopography are all spatially auto-correlated within 100, 87 and 53 m, respectively. Bivariate Moran's I revealed a positive but weak spatial correlation between SOM and Ks and a moderately strong negative spatial correlation between SOM and soil surface microtopography. A map of SOM was generated using simple kriging, which predicts higher SOM in the center of the ecosystem, at lower elevations; and lower SOM at the edges of the study area, at higher elevations. Local depressions in the center of the ecosystem provide a wetter and therefore more anaerobic environment, thereby decreasing carbon mineralization rates and enabling peat accumulation. The low hydraulic conductivity of the degraded peat in the presence of lower micro-elevations in the center of the ecosystem is likely to increase the residence time of floodwater and thus may enhance (new) peat accumulation. Thus, we conclude that, for the restoration of non-tidal coastal mires where flooding events are not as frequent, K_s and soil surface microtopography are even more important factors to consider than for tidal systems.

Keywords: hydraulic conductivity, peatland, restoration, soil organic matter, spatial variability

4.1. Introduction

Mires and peatlands account for only 3 % of the global land surface (Yu et al., 2010), primarily occurring in boreal and temperate regions, with a smaller proportion in the tropics. Nevertheless, they may store of up to 644 Gt of carbon (Dargie et al., 2017; Page et al., 2011; Yu et al., 2010) or about 21 % of the global total soil organic carbon stock of 3000 Gt (Leifeld & Menichetti, 2018; Scharlemann et al., 2014). Carbon sequestration and greenhouse gas emissions avoidance through peatland restoration have been recognized as climate change mitigation strategies. Furthermore, at present, human activities are either draining or mining about 12 % of global peatlands (Joosten, 2010), thereby changing them from long-term carbons sinks into sources (Leifeld & Menichetti, 2018) by accelerating the carbon mineralization process of soil organic matter (Brandyk et al., 2002).

Mires and peatlands that are located in low-lying coastal areas are of particular interest, as coastal wetlands sustain the highest rates of carbon sequestration per unit area of all ecosystems (Rogers et al., 2019). In low-lying coastal areas, peatlands form by the accumulation of organic material over millennia and are often regarded as the interface between the land and the sea. While there is a large uncertainty in terms of the total land area of coastal peatlands (Henman & Poulter, 2008), analysis by Chmura et al. (2003) reveal that saline wetland soils (including salt marshes and mangrove swamps) store more than 10,000 Tg of carbon. Nevertheless, coastal peatlands face additional threats from climate change as rising global sea levels may drive future releases of stored carbon (Henman & Poulter, 2008; Whittle & Gallego-Sala, 2016).

Drainage of peatlands can alter hydro-physical properties of peat soils such as soil organic matter content (Heller & Zeitz, 2012), pore structure and hydraulic conductivity (Liu et al., 2016; Rezanezhad et al., 2016; Zeitz & Velty, 2002) and consequently may alter hydrological processes (Holden et al., 2006; Holden & Burt, 2003) as well as change water chemistry (Holden et al., 2004) and vegetation composition (Schrautzer et al., 2013). As hydraulic properties control soil moisture, they in turn drive carbon and nitrogen dynamics (Kluge et al., 2008). For example, under water-saturated conditions, the low oxygen available limits microbial activity and CO2 fluxes (Säurich et al., 2019). Denitrification is limited by water availability while nitrification is limited by aeration (Säurich et al., 2019). Thus, when there is a lack of soil moisture, aeration of the peat and subsequent mineralization and nitrification of organic nitrogen releases large amounts of nitrates (Holden et al., 2004; Tiemeyer & Kahle, 2014).

Soil surface microtopography, which is the micro-elevation of the surface of the soil (Z. Li & Z. Chen, 2012), can have a large influence on flow processes at the soil surface (Fox et al., 1998; van der Ploeg et al., 2012). Microtopographic variation translates into differences in hydrology within a wetland, with topsoil and vegetation being drier at higher elevations than in depressions because of increased distance from the water table (Benscoter et al., 2005). Therefore, microtopography can give a good insight into the wetness of a given area, influenced by both groundwater and surface water dynamics, and thus can be associated with spatial patterns in soil organic matter content (SOM). For example, Zheng et al. (2019) showed that there is less SOM at higher elevations because it decomposes faster under aerobic conditions than under anaerobic conditions.

Soil organic matter content and bulk density are generally negatively correlated both in mineral soils (Adams, 1973; Liu & Lennartz, 2019a; Perie & Ouimet, 2008; Rawls, 2004) and in organic soils (Adams, 1973; Liu & Lennartz, 2019a; Perie & Ouimet, 2008). Additionally, SOM and total porosity are positively correlated (Grover & Baldock, 2013; Kechavarzi et al., 2010; Lennartz & Liu, 2019). Furthermore, SOM and K_s have been found to be positively correlated according to Zare et al. (2010), Nath and Krishna (2014) and Zhang et al. (2018) for mineral soils and by Boelter (1969) and Liu and Lennartz (2019a) for peat soils.

Although it is well acknowledged that physical soil properties and hydraulic parameters have spatial dependencies (Bevington et al., 2016), our understanding of spatial variability of hydrophysical properties of organic soils is limited compared to that of mineral soils (Lewis et al., 2012). There is a wealth of studies which predict mineral soil properties as part of the digital soil mapping literature (Ma et al., 2019), some of which focus on peat (Minasny et al., 2019), although most of these studies focus on much larger scales (e.g. subnational, national, regional, global) than that of the present study. Digital mapping methodology combines field observations with factors that are known to affect soil properties. For example, Rudiyanto et al. (2018) utilized digital elevation models, geographical information and radar images, along with machine learning models, to derive spatial prediction functions and map peat thickness on an island in Indonesia. Kriging methods have also been used in several studies to predict peat thickness (Altdorff et al., 2016; Beilman et al., 2008) and volume (Jaenicke et al., 2008).

Microtopography is known to affect soil hydrophysical properties in mire ecosystems. A study conducted by Baird et al. (2016) found clear patterns in K_s between adjacent hummocks and hollows (microforms) at 0.5 metre depth in a raised bog. Morris et al. (2019) also explored the effect of microforms on K_s (vertical and horizontal), and collected samples from hummocks and lawns in a raised bog. They found a strong independent influence of microhabitat on log-transformed vertical K_s. Similarly, Branham and Strack (2014) found K_s to be higher in hummocks than in the hollows at the surface of a Sphagnum-dominated fen and bog. Such relationships between microtopography and soil hydrophysical properties have been generally explored in raised and blanket bogs with little to no focus on other mire systems. Thus, there is a gap in our understanding of such relationships in degraded coastal mires following rewetting.

Therefore, the objectives of our study are to (1) understand spatial structures of hydrophysical properties of organic soils, (2) investigate spatial patterns of organic matter accumulation in relation to microtopography, and (3) understand the role of the organic horizon with respect to underlying mineral soil with respect to hydrological connectivity, in a coastal flood mire. We also use spatial measurements of hydrophysical properties (K_s and SOM) and soil surface microelevation to make spatial predictions, which has not been done in coastal mires.

4.2. Methods

4.2.1. Study site

The study site lies in the north-eastern German federal state of Mecklenburg-Western Pomerania which is home to around 3000 km^2 of peatlands (13 % of the total land area of the state; Tiemeyer et al., 2006). The study site is part of a non-tidal coastal flood mire known as "Karrendorfer Wiesen" and is located (54.1576° N, 13.3860° E) between Greifswald and Stralsund, on a peninsula in the Baltic Sea (**Figure 4.1**). Karrendorfer Wiesen has an area of approximately 3.5 km^2 and is a part of the 400 km^2 of coastal peatlands covering the state (Jurasinski et al., 2018). It is characterized by weakly undulating ground moraine, which was flooded during postglacial transgressions. Currently it lies within the natural flooding zone of the “Nordmecklenburgschen Bodden” (Bernhardt & Koch, 2003). The study site was drained in 1820. In 1850, a dike system was constructed and the area was used intensively as cropland and pasture. The height of the dike was increased between 1971 and 1974. The dike blocked the flow of seawater flooding to the mires, reducing soil salinity; and the accomplished drainage lowered the water table, accelerating organic matter mineralization. However, in the early 1990s the area lost its importance as pasture due to a small number of landowners coupled with the poor condition of the old dike system (Lampe & Wolrab, 1996). Thus, in 1993 the old dikes were removed and a new dike system was built so as to reinitiate natural flooding dynamics on part of the area (Beyer et al., 2019). Currently this area is used as a low-intensity pasture (Bernhardt & Koch, 2003; Beyer et al., 2019). Karrendorfer Wiesen is recognised as a “National Natural Heritage” by the Federal Government of Germany, and is therefore protected for nature conservation. It is also an important coastal bird sanctuary (Janssen et al., 2019).

The coastal mire is represented by a mosaic of micro-elevational changes consisting of marly till and sandy soils, and interspersed low-lying areas (Bernhardt & Koch, 2003) consisting of fen gley soils with 13–28 cm of peat (Janssen et al., 2019). Peat is an organic soil which is composed of partially decomposed plants (Kelly et al., 2017). According to Rydin and Jeglum (2006) there is no general agreement on how to define peat using organic matter content and its defining range may vary from 20 % to 80 % organic matter by weight. For this reason, coupled with the fact that there is high variance of the percentage of organic matter found in our samples, throughout this article we refer to the soils of this coastal mire as “organic soil” or “peat”. The pH of the soil ranges from 4.4 to 6.1 depending on the depth. The soils have a bulk density of around 0.57 g cm^{-3} with a total porosity of 0.71 cm^3 cm^{-3} (Liu & Lennartz, 2019b).

According to vegetation data collected by Beyer et al. (2019), the study site is characterized by salt marsh species. *Agrostis stolonifera* is the most dominant species followed by *Alopecurus geniculatus* and *Juncus gerardii*. *Elymus repens, Triglochin martima, Deschampsia cespitosa, Trifolium repens* and *Trifolium hybridum* occur sporadically while *Spergularia salina, Poa pratensis, Potentilla anserina, Aster tripolium* and *Plantago maritima* appear less often throughout the study area.

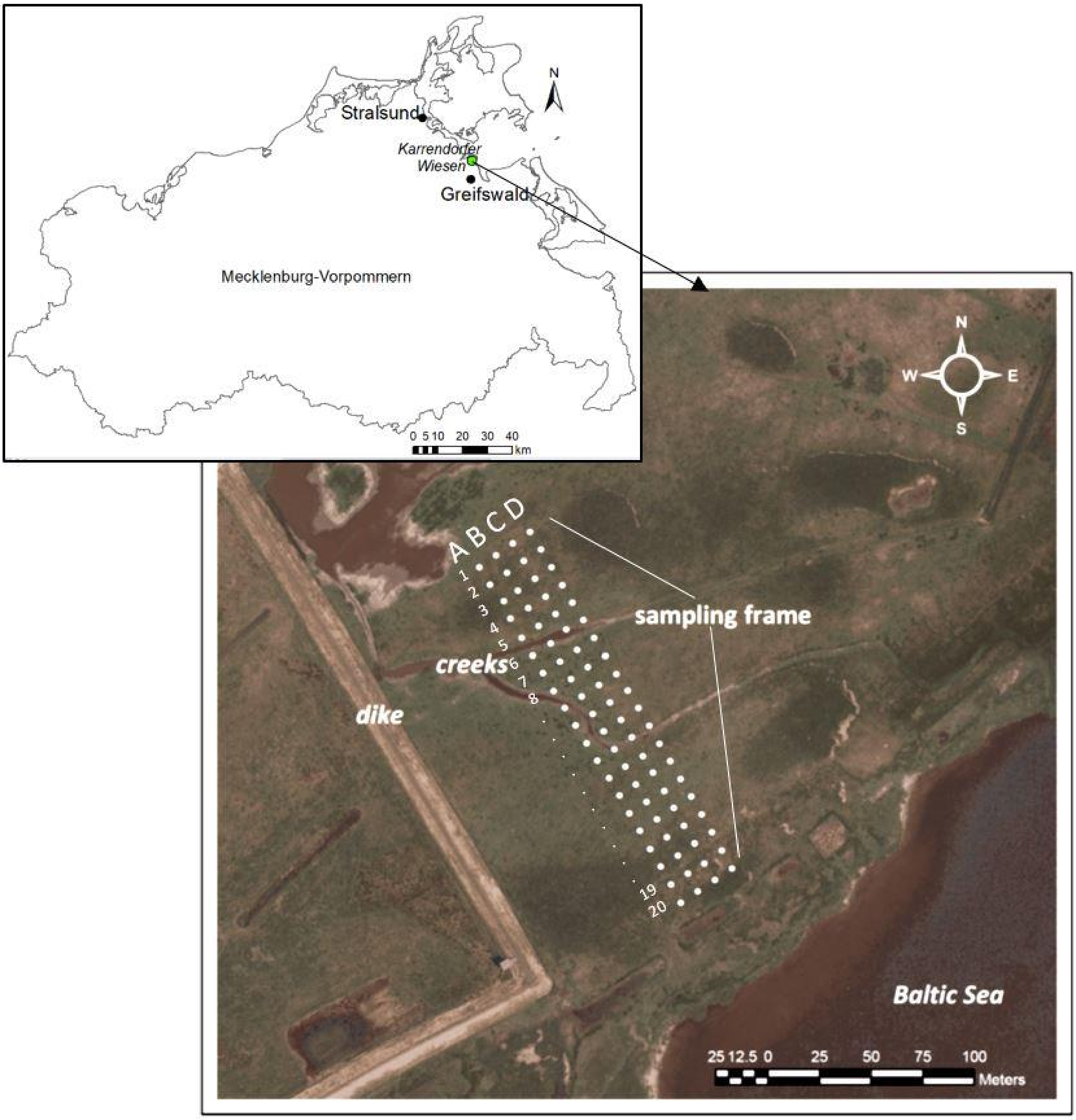


Figure 4.1. Location of the study area "Karrendorfer Wiesen". The white dots on the map show the locations where soil samples were collected.

4.2.2. Soil sample collection and saturated hydraulic conductivity determination

The sampling frame covers an area of 6000 m2 (200 × 30 m). For determining soil organic matter content (SOM), soil samples at a depth of about 15 cm were collected using a gouge auger every 10 meters at 80 points. Prior to soil sample collection, at the same depth, in situ saturated hydraulic conductivity (K_s) was measured using a direct-push piezometer with falling head which has been used for peat soils in previous studies (Mustamo et al., 2016; Postila et al., 2015; Ronkanen et al., 2005; Saarinen et al., 2013). This device is particularly useful for our study area because laboratory based K_s measurements would require the collection of large amounts of undisturbed soil which is not possible given the protection status of the Karrendorfer Wiesen. Other field methods for measuring K_s such as the piezometer slug test would require extensive installation of piezometer pipes, which is again not feasible given the protection status of the study site. However, K_s could be measured at only 39 points, as it was not possible to penetrate the soil surface at all locations using the device.

The direct-push piezometer (see **Figure 4.2**) is inserted into the soil at the desired depth slowly and without any twisting motion to avoid potential smearing. Additionally, a tripod with a clip is used to hold the device in place. Afterwards, water collected from the field site is poured into the reservoir. The hydraulic head is noted after which the control valve at the base of the reservoir is released to allow the water to flow through the pipe. At the tip of the piezometer, there is a meshed opening (the screen) on two sides with a diameter of 2 cm. It is here that the piezometer water comes into contact with the soil. The falling head is timed and recorded. Due to loss of head in the piezometer the method allows for accurate Ks measurements below 0.002 m s^{-1} (Ronkanen et al., 2005). Prior to carrying out fieldwork, several in situ measurements of K_s using the direct-push piezometer were compared to several laboratory measurements using a constant-head upward-flow permeameter (Liu et al., 2016) on undisturbed soil samples collected from the same locations (see **Figure SM 4.1).** Results from both methods are statistically similar.

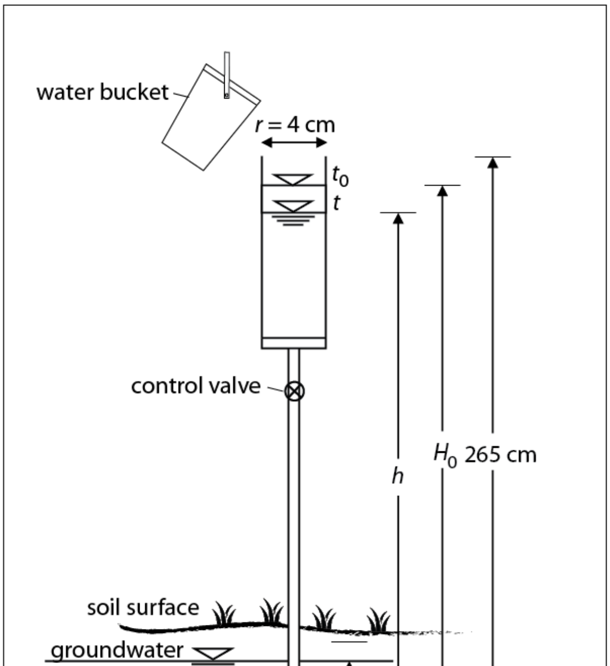


Figure 4.2 A schematic diagram of a direct-push piezometer with falling head (adapted from Ronkanen & Klove, 2005). Height from the centre of the outlet screen = 265 cm; r (inner radius of the reservoir) = 4.0 cm; length and diameter of the outlet screen = 1.2 cm; H = water level from outflow point of piezometer; H0 = the initial water level in the reservoir, h = water level in the container at a time point t. The tip of the device is sharp and connected smoothly to the body of the device to avoid compression of soil when inserting into the ground.

The rate of the outflow (q) at the piezometer screen/outlet at any time (t) is proportional to the hydraulic conductivity (K) of the soil and to the unrecovered head difference (H - h) (Hvorslev, 1951; Mustamo et al., 2016), so that:

$$q(t) = \pi r^2 \frac{\partial h}{\partial t} = FK\ (H - h) \qquad \textbf{(4.1)}$$

where r = radius of the piezometer reservoir (m), H = water level from outflow point of piezometer (m), h = water level in the reservoir (m), F = 9 (dimensionless), shape factor calculated according to equation provided in Akanegbu (2013), t = time (s).

From **Equation 4.1** the following equation may be derived:

$$ln\left(\frac{h-H}{H_0-H}\right) = -\ \frac{FK}{\pi r^2} t \qquad \textbf{(4.2)}$$

A plot of the left-hand side of **Equation 4.2** against time represents a straight line on a semi-logarithmic graph. Therefore, K can be calculated from the slope of this straight line.

Due to the shallow depth of peat (13 – 28 cm), it is important to understand the role of the underlying mineral horizon in terms of groundwater flow. Therefore, further soil samples were collected from three locations (B2, B9, and B20; see **Figure 4.1**) at depths of 20-40 cm, 40-60 cm, and 60-80 cm for textural analysis.

4.2.3. Laboratory analysis

Soil organic matter content (SOM) was determined by loss-on-ignition according to DIN 18128 (DIN 2002) and expressed as weight percent (%w/w). Advantages of this method are (1) a large number of samples can be run simultaneously and (2) equipment cost is low (De Vos et al., 2005). Another prominent method of determining SOM is the Walkley-Black acid digestion method. However, for soils with high organic matter content this method may result in inaccuracies due to incomplete oxidation of organic carbon in the sample (Lefèvre et al., 2017).

Additionally, particle size distribution (soil texture) from locations B2, B9 and B20 and three depths of the mineral horizons (20–40, 40–60 and 60–80 cm) was determined using the sieving and sedimentation method according to DIN ISO 11277 (DIN ISO, 2002). Prior to soil texture analysis, all samples were tested for carbonates by using 10 % HCl which did not result in any fizzing indicating a lack of significant amounts of carbonates. SEDIMAT 4-12 (UGT), which works on the basis of the KÖHN analysis to DIN ISO 11277, was used to determine the three silt fractions (coarse, medium, fine) and one clay fraction. The remaining sand fractions were further separated using sieves with mesh sizes of 63 µm, 200 µm, and 630 µm.

4.2.4. Elevation data and geostatistical analysis

All sampling locations and corresponding soil surface elevations were recorded using a high precision GNSS receiver (Leica Viva GS08 plus) which uses real-time kinematic positioning. ArcMap 10.5.1 was used to analyze spatial data. Using the Geostatistical Wizard, accessed through Geostatistical Analyst extension, empirical variograms of Ks (log-transformed), SOM and soil surface microtopography (SSM) were generated. Empirical variograms of K_s (log-transformed) and SSM were fitted with Gaussian models, while the variogram of SOM was fitted with a "Stable" model with parameter = 1.898. These variograms were then utilized to generate prediction maps using "simple kriging" method. For calculation of partial sill and nugget, weighted least squares was used. The models and the parameters are described in the Results section (see **Table 4.3**).

A "leave-one-out" method was used for cross-validation. Each data location is removed, one at a time, and the associated data value is predicted. The predicted and actual values at the location of a removed point are then compared and this procedure is repeated for a second point, a third and so on. All the measured and predicted values were compared using scatterplots and quantile-quantile plots. Plots of predicted versus observed values of all three variables (K_s, SOM, and SSM) are provided in **Figure SM 4.2.** The maps of prediction standard errors for all three kriged variables are presented in **Figure SM 4.3**.

For understanding the association between observed SOM and K_s and between SOM and SSM, in addition to calculating the conventional Pearson correlation, bivariate Moran's I was computed using GeoDa version 1.12, as it is more suitable for variables with spatial dependencies. Lee (2017) states that bivariate spatial dependence refers to circumstances in which observational units in close proximity hold shared information in terms of their bivariate association, and this violates the assumption of independent sampling. Thus, the shared information spuriously strengthens or weakens the nature of correlation between the two variables under investigation, thereby making any conventional statistical inferences considerably questionable.

Moran's I is a well-known indicator of spatial autocorrelation. Moran's I values range from -1 to 1. A '0' value indicates no spatial autocorrelation (i.e. perfect spatial randomness). A '-1' suggests perfect negative spatial autocorrelation or clustering of dissimilar values (i.e. perfect dispersion) while +1 indicates perfect clustering of similar values or, in other words, high values or low values cluster together (Tu & Xia, 2008). Similar to the univariate Moran's I, the bivariate Moran's I can help to understand spatial dependencies but it helps to assess such relationships between two variables instead of one. The bivariate Moran's I can be visualized as the slope in a scatterplot of the spatially lagged values of one variable (e.g. soil organic matter) on the second variable (e.g. soil surface microtopography). If the slope of this scatterplot is significantly different from zero, then there is a bivariate spatial relationship between the two variables (Sridharan et al., 2007). The test is based on an assumption of constant mean and a constant variance (Anselin, 2019).

4.3. Results

4.3.1. Hydrophysical soil properties and soil surface microtopography

The K_s values of investigated soils ranged from 5.56×10^{-9} m s^{-1} to 4.64×10^{-7} m s^{-1}, and had the highest coefficient of variation (CV) = 199.99 %, while the soil organic matter content (SOM) varied from 1.36 to 59.29 wt% with a CV of 56.65 %. SSM had a very low CV of around 24 % and ranged from 223 cm to 918 cm above mean sea level **(Table 4.1**).

Table 4.1 Summary statistics for hydraulic conductivity (K_s), soil organic matter (SOM) and soil surface microtopography (SSM).

Soil variables	Mean	SD	CV (%)	Minimum	Maximum	*n*
K_s (m s^{-1})	4.17×10^{-8}	8.33×10^{-8}	199.99	5.56×10^{-9}	4.64×10^{-7}	39
SOM (%w/w)	21.36	12.10	56.65	1.36	59.29	80
SSM (m above MSL*)	0.57	0.14	24.56	0.22	0.92	80

**MSL = mean sea level.*

4.3.2. Soil texture of the underlying mineral horizon

Soil texture analysis of the mineral soils underlying the peat horizon (0–20 cm) at three locations reveals that for almost all the locations and all depths, the soil can be classified as sandy loam (following USDA) and as medium loamy sand according to German soil textural classification (Eckelmann et al., 2006). **Table 4.2** provides the detailed results of the texture analysis.

Table 4.2 Soil texture classes according to German classification (Eckelmann et al., 2006). The English terms for the German soil classes were used after Bormann (2007).

Location	Depth (cm)	Clay	Silt	Sand	Soil Texture Class (Germany)
B2	20-40	6.80	37.57	55.63	medium silty sand
	40-60	9.26	33.44	57.30	medium loamy sand
	60-80	16.65	28.85	54.50	highly loamy sand
B9	20-40	12.67	36.25	51.08	highly loamy sand
	40-60	5.59	24.70	69.70	slightly loamy sand
	60-80	10.36	23.95	65.69	medium loamy sand
B20	20-40	10.01	27.94	62.04	medium loamy sand
	40-60	11.14	27.31	61.55	medium loamy sand
	60-80	8.87	20.89	70.24	medium loamy sand

4.3.3. Spatial structure of hydro-physical soil properties and surface microtopography

Variograms provide a measure of the spatial dependence of soil properties. Both soil properties as well as SSM can be observed to be spatially dependent. While log (K_s) follows a Gaussian curve (R^2 = 84.85), SOM follows a Stable model with a parameter of 1.898 (R^2 = 90.93). The semivariance of log (K_s) increases initially and then levels out at a lag distance of 87 m while SOM levels off at a slightly higher lag distance of 100 m (see **Figure 4.3**). Therefore, beyond these separation distances the hydro-physical soil properties under investigation are not auto-correlated. Soil surface microtopography (SSM) follows a Gaussian curve (R^2 = 93.64), with the semivariance increasing until the range of 53 m and afterwards levels out.

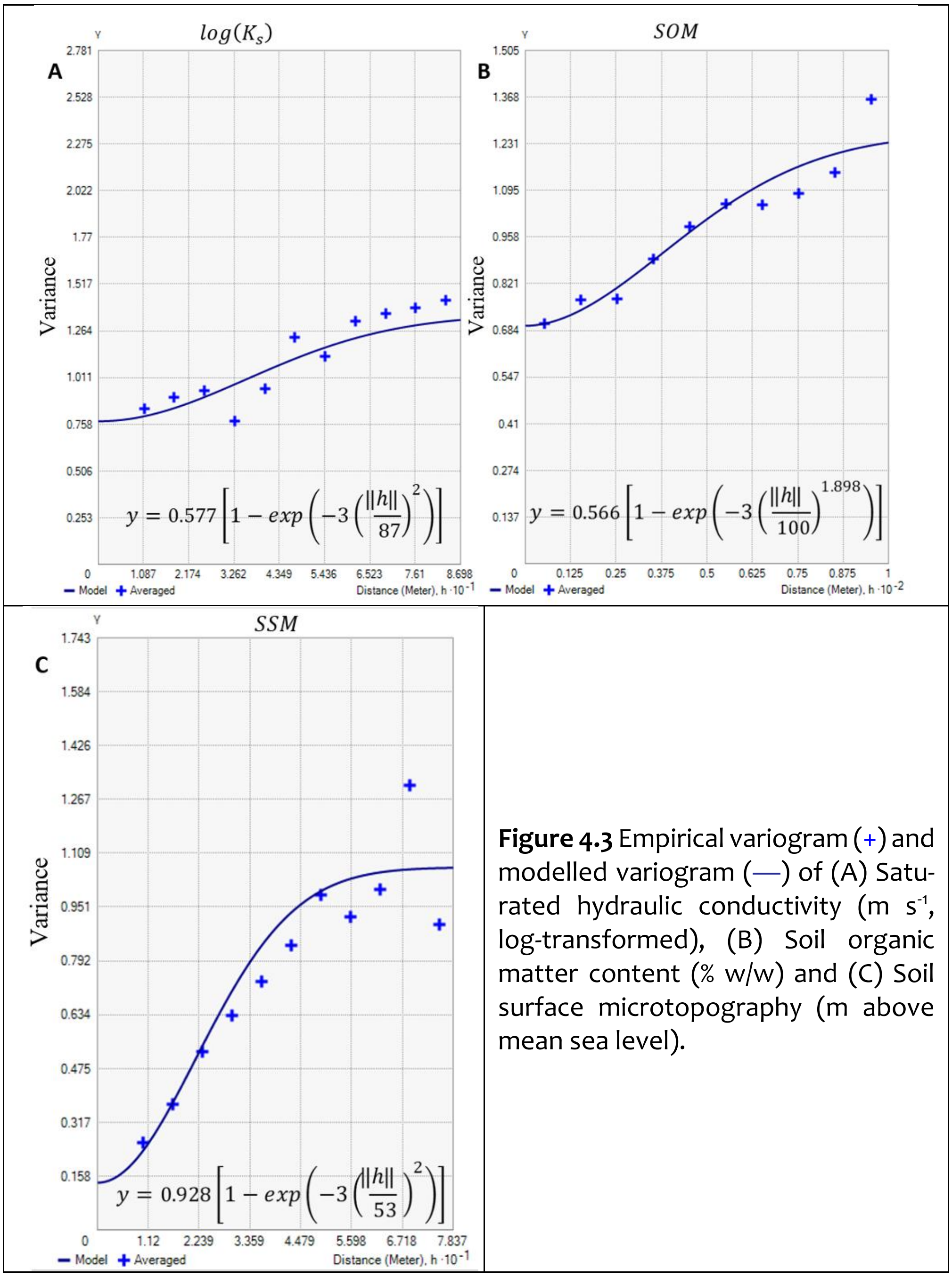


Figure 4.3 Empirical variogram (+) and modelled variogram (—) of (A) Saturated hydraulic conductivity (m s^{-1}, log-transformed), (B) Soil organic matter content (% w/w) and (C) Soil surface microtopography (m above mean sea level).

The nugget to sill ratio (NSR) of 57.3 %, 55.2 % for K_s and SOM, respectively, indicates only a moderate spatial autocorrelation (following Cambardella et al., 1994) for both soil properties. However, for SSM, a NSR of about 13 % indicates strong spatial autocorrelation (**Table 4.3**). Thus, SSM is more strongly autocorrelated than K_s and SOM.

Table 4.3. Variogram models and parameters for SOM and log(K_s).

Soil Properties	Model	Nugget (C_0)	Sill (C_0+C)	Partial Sill (C)	C_0/ C_0+C (%)	Range (m)	R^2 (%)
Log(K_s)	Gaussian	0.775	1.352	0.577	57.322	87	84.85
SOM	Stable (Parameter= 1.898)	0.698	1.264	0.566	55.222	100	90.93
SSM	Gaussian	0.139	1.067	0.928	13.027	53	93.64

Ideally, at zero separation distance (h) the variance should also be zero. However, many soil properties have non-zero variances as h tends to zero (Trangmar et al., 1985). The variance at zero lag distance is called the nugget effect which represents the local variation occurring at scales finer than the sampling interval, such as those due to sampling error, fine-scale spatial variability and the measurement error. Log (K_s) has a slightly higher nugget effect (0.775) compared to that of SOM (0.698), while SSM has a much smaller nugget effect (0.139) indicating much lower errors and lower fine-scale spatial variability.

4.3.4. Spatial heterogeneity of hydraulic conductivity, SOM and soil surface microtopography

There is a general tendency for K_s to increase in the direction of the sea, with no clear tendencies closer to the inland water bodies (**Figure 4.4A**). However, the plot of predicted K_s versus observed K_s shows that predictions of K_s have substantial errors, especially in comparison to the plots of SOM and SSM (see **Figure SM4.2**). It can be observed that there is higher SOM closer to the center, while much lower SOM at the West and South East ends of the study area (**Figure 4.4B**). The SSM map shows the highest elevation class (0.76–0.92 m) at the edges (West and South East of the map) and the lowest elevation classes (0.22–0.37 m) at or around the center (**Figure 4.4C**). Therefore, the spatial distribution of SOM can be described by the spatial variation in microtopography in the study area. This can be further illustrated by the bivariate association between the two variables (see **Figure 4.5**), discussed in the next subsection.

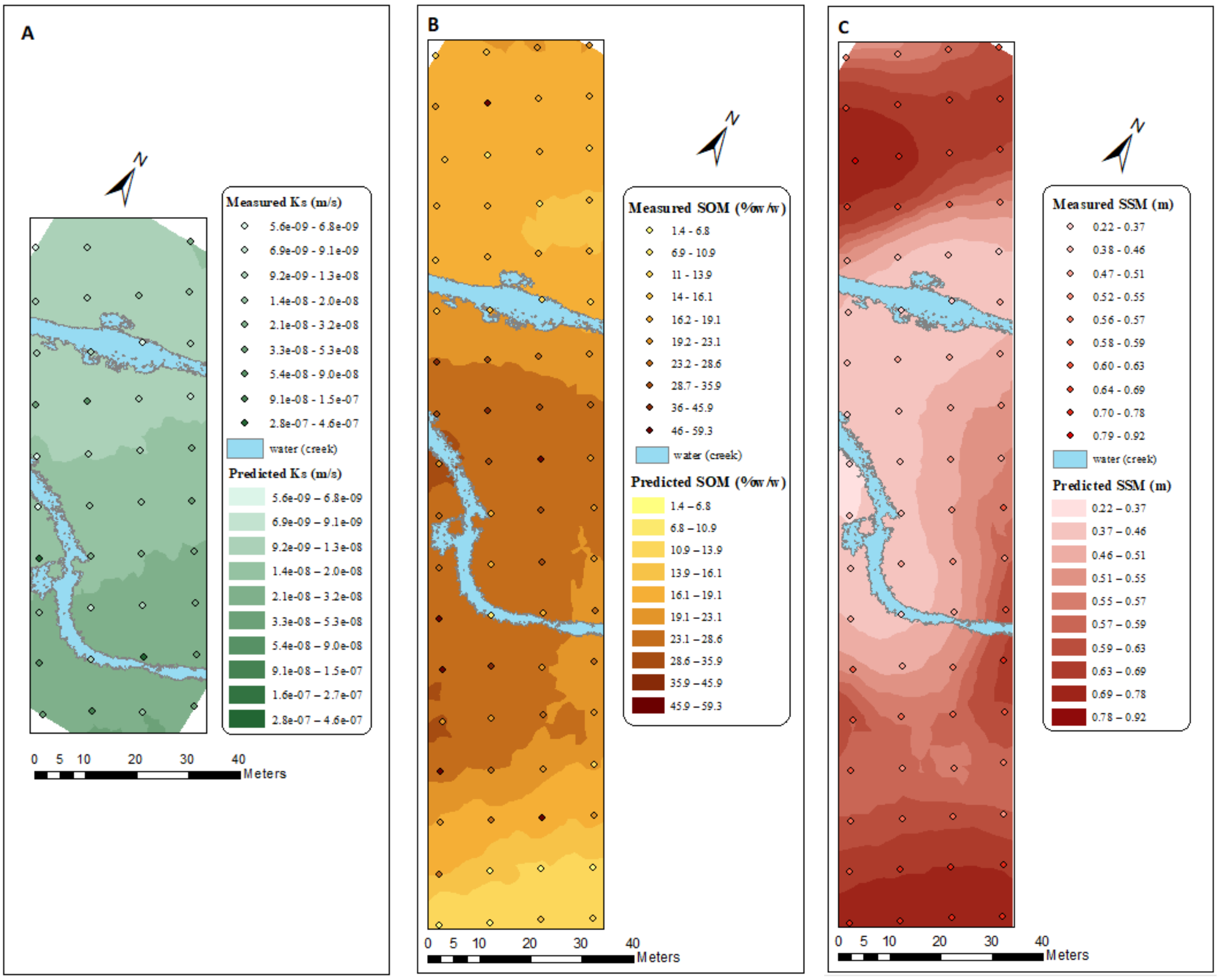


Figure 4.4 Contour maps showing spatial distribution of **(A)** K_s (m s^{-1}) **(B)** SOM (%w/w) **(C)** SSM (in metres above mean sea level).

4.3.5. Bivariate spatial dependencies

A significantly moderate and positive Pearson correlation was obtained between log(Ks) and SOM ($r = 0.53$, $p = 0.0008$). In terms of bivariate spatial autocorrelation between the two variables, a positive Moran's I was found (Moran's I = 0.1440) which, although low, is still significant (pseudo-p = 0.02, permutations = 999, z = 2.1870).

A negative spatial autocorrelation (Moran's I = -0.2993) between SOM and SSM were found to be significant (pseudo p-value = 0.001, permutations = 999, z = -5.67, weights generated using Queen's contiguity; see Figure 5). This means that both SOM and SSM are significantly and negatively spatially autocorrelated - a decrease in soil surface elevation is associated with an increase in SOM across space. The possible underlying mechanism behind this is discussed in the following section.

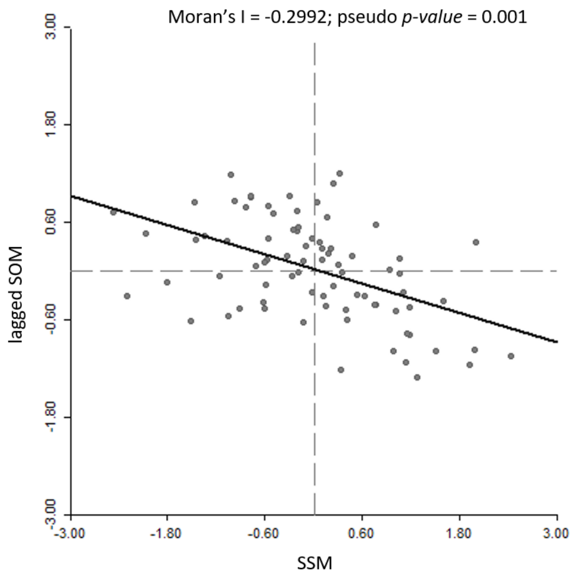


Figure 4.5 Bivariate Moran's I showing the spatial relationship between observed soil surface microtopography (micro-elevation) and observed soil organic matter content (lagged).

4.4. Discussion

Compared to mineral soils, organic soils of coastal flood mires are less studied, especially in relation to space and microtopography. Our study investigates existing spatial structures and patterns of soil hydrophysical properties such as SOM and K_s and how microtopography may affect such properties. Understanding spatial patterns of the accumulation of SOM and distribution of Ks and how they are modified by surface microtopography in such mire ecosystems are of particular interest since formation of peat and maintaining hydrological connectivity are prerequisites for the effective restoration of mire and peatland ecosystems.

Analysis of SOM of the topsoil (organic horizon) reveals that the coastal flood mire has a mean SOM of 21.4 % (SD = 12.1 %), ranging from 14 % to around 60 %, with a CV of 57 % which is indicative of a very high variation following Warrick (1980) and Paz Ferreiro et al. (2016). For mineral soils, Bernardi et al. (2017) reports a substantially lower CV for SOM, of around 15 %, while Paz Ferreiro et al. (2016) reports CV ranging from 27 % to 40 % (depending on soil depth and land use). For peat soils of a drained wetland located in the Qinghai-Tibet plateau of China, Bai et al. (2010) reports a CV of 9 % for soil organic carbon which is a substantially lower variation than that of the current study. Land use can also have an effect on the variation of soil organic matter in peat soils with CV ranging between 109 % in arable peat, 68 % in woodland peat, 45 % in grassland peat and 38 % in moorland peat according to a study carried out in south-west England (Glendell et al., 2014).

For the organic horizon of the study site, the mean K_s is 4 × 10-8 m s-1 (SD = 8×10^{-8} m s^{-1}) with a minimum of 6×10^{-9} m s^{-1} to a maximum of 5×10^{-7} m s^{-1}. Our values of K_s are very low, about 2 to 4 orders of magnitude lower than that estimated by van Dijk et al. (2017) for peat sediments in a coastal wetland of the Netherlands (around 7×10^{-5} m s^{-1}). K_s shows a very high variation, with a

CV of about 200 percent which is consistent with values reported for peat soils (CV = 282 %, calculated from a meta-analytical study by Liu & Lennartz, 2019a).

Soil organic matter content and K_s (log-transformed) were found to be significantly and positively correlated, but only moderately ($r = 0.53$, $p = 0.0008$). A positive correlation as such is consistent with the results of several other studies, including those for mineral soils (Hur et al., 2009; Nath & Krishna, 2014; Zare et al., 2010) as well as for organic soils (Lennartz & Liu, 2019; Liu & Lennartz, 2019a). Macroporosity is a major factor controlling K_s (Liu et al., 2016). Liu and Lennartz (2019a)found that macroporosity is higher in peat with high organic matter content than in soils with low organic matter content.

In terms of spatial dependencies in our study, K_s and SOM are spatially autocorrelated with each other with a significant and positive Moran's I (Moran's I = 0.1440, $p = 0.02$), suggesting non-randomness in their overall spatial pattern. Both K_s and SOM show moderate spatial autocorrelation as defined by Cambardella et al. (1994), with a range of about 87 m for K_s and 100 m for SOM. For mineral soils Zeleke and Bing (2005) report a much lower spatial range of about 50 m for K_s and 43 m for organic carbon. Paz Ferreiro et al. (2016) investigated the spatial variability of mineral soil properties according to different land uses and depth, and also found much lower spatial ranges with a minimum of around 12 m to a maximum of about 60 m, depending on land use and soil depth. However, the predicted map of K_s should be interpreted with caution, as the plot of predicted values against observed values show poor prediction performance which may be attributable to (1) small sample size ($n = 39$) and (2) large sample intervals (10 m). This was not the case for the prediction of SOM, which performed much better, while SSM prediction performed the best among all three variables. This may be an indication that K_s follows spatial patterns at much smaller scales than SOM and SSM in coastal mires as it has also been observed from the variograms (**Figure 4.3**). Although a sampling interval of 10 m allowed us to have larger spatial coverage, it may have resulted in an increased nugget effect especially for the prediction of K_s. Follow-up studies should incorporate nested sampling of different interval lengths, to understand multiple scales at which soil physical properties and soil surface microtopography interact in coastal mires.

A key finding of our research is that SOM and soil-surface microtopography (SSM, micro-elevation) are significantly negatively autocorrelated (Moran's I = 0.2993, $p = 0.001$), which can also be visually observed by comparing the kriged map of SOM with the map of SSM. The SOM map predicts higher organic matter content in the center of the ecosystem at lower elevations, while at the edges of the study area, at higher elevations, SOM decreases. Local depressions in the center of the ecosystem provide a wetter and therefore a more anaerobic environment as oxygen diffuses 10,000 times slower through water than it does through air and in the presence of waterlogged decomposing plant material the supply of oxygen is rapidly depleted (Clymo, 1983; Lindsay & Andersen, 2016). Thus, under anaerobic conditions, carbon mineralization rate decreases, enabling the accumulation of organic matter, as has been confirmed by a multitude of studies (Aerts & Ludwig, 1997; Benavides, 2015; Blodau et al., 2004; Kettunen et al., 1999; Öquist & Sundh, 1998; Yavitt et al., 1997). However, unlike our study, most of these studies were based on laboratory experiments and not on field research. We may therefore generalize that SSM is an important feature to take into account while planning restoration measures especially when there is lack of water table monitoring with high spatial resolution and when addressing coastal mires with low K_s, located at the Baltic Sea coast - where soils are not subjected to significant tidal flooding.

For almost all three locations and for all depths (of the underlying mineral horizons), the soil can be characterized as sandy loam (USDA classification) which, according to Schaap et al. (2001), has an average K_s of about 4×10^{-6}ms^{-1} (SD = 5×10^{-7} ms^{-1}). Therefore, even the maximum *in situ* K_s (5×10^{-8} m s^{-1}) of the organic horizon is two orders of magnitude lower than that of the underlying mineral horizon, which is an indication that the organic horizon with its relatively low hydraulic conductivity acts as a hydrological barrier, at least in terms of infiltration. This finding is consistent with a study carried out on a different Baltic coastal peatland which found that the peat layer has K_s ranging from 1×10^{-6} to 1×10^{-8} m s^{-1} which is one to two orders of magnitude lower than that of the underlying mineral soil with a K_s of 2×10^{-5} to 6×10^{-5} m s^{-1} (Ibenthal, 2019). This brings to the fore the question of whether the organic soil or peat horizon of other coastal wetlands which developed under similar conditions also acts as a hydrological barrier.

In terms of rewetting, it is therefore useful to know locations with higher microelevations because these are potential hotspots for faster degradation rates as flooding or rainfall events may lead to water accumulating in areas of low micro-elevation only, through overland flow or inflow through the existing creek system. Furthermore, during rainfall events which lead to a rise in groundwater level, areas with lower elevations will become saturated first, and only afterwards the water table may reach higher elevations. In situations where restoration projects utilize water level monitoring wells, data on micro-elevation may inform locations where installations should be set up. In addition, the low hydraulic conductivity of the degraded peat in the presence of lower micro-elevations in the center of the ecosystem is likely to increase the residence time of floodwater and thus may enable (new) peat accumulation. Thus, we conclude that for the restoration of non-tidal coastal mires, where flooding events are not as frequent, K_s and SSM are even more important factors to consider than for tidal systems. Extensive research in such under-studied and complex ecosystems is required in order to better understand the underlying mechanisms of peat formation following dike removal (rewetting).

Author Contributions

HL, **SA** and **BL** conceived the research idea and design. **SA** undertook the field measurements and laboratory analysis. **BK** provided the direct-push piezometer. **SA** analyzed the data and designed the Figures with input from **FB**. **SA** wrote the first draft while all authors discussed the results and revised the manuscript.

Acknowledgments

The authors thank Johanna Blossei, Miaorun Wang Tilo Hartwig and Laurenz Teuber for helping with data collection and logistics; Evelyn Bolzmann for providing guidance on the laboratory analysis; Matthias Naumann for providing support with high precision GPS; Görres Grenzdörffer, Donald Myers and Shajratul Alam for their valuable advice on geospatial analysis; and Lennart Gosch for German language support. The European Social Fund (ESF) and the Ministry of Education, Science and Culture of Mecklenburg-Western Pomerania funded this work within the project WETSCAPES (ESF/14-BM-A55-0028/16). Financial support was also received from the Research Training Group Baltic TRANSCOAST program, funded by the DFG (Deutsche Forschungsgemeinschaft) under Grant Number DFG-GRK 2000/1.

Supplementary Material to Chapter 4

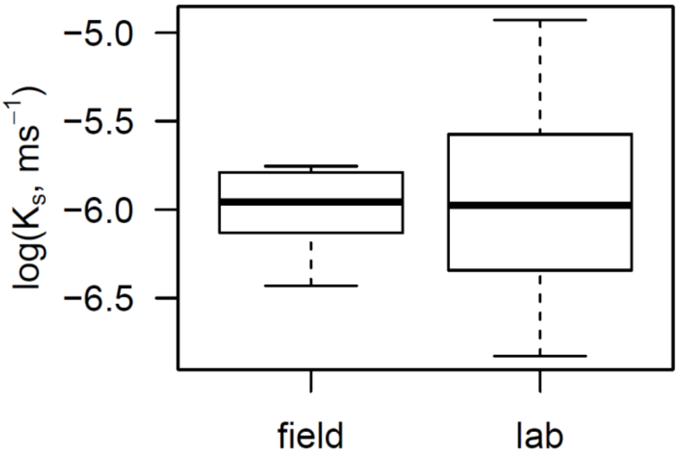
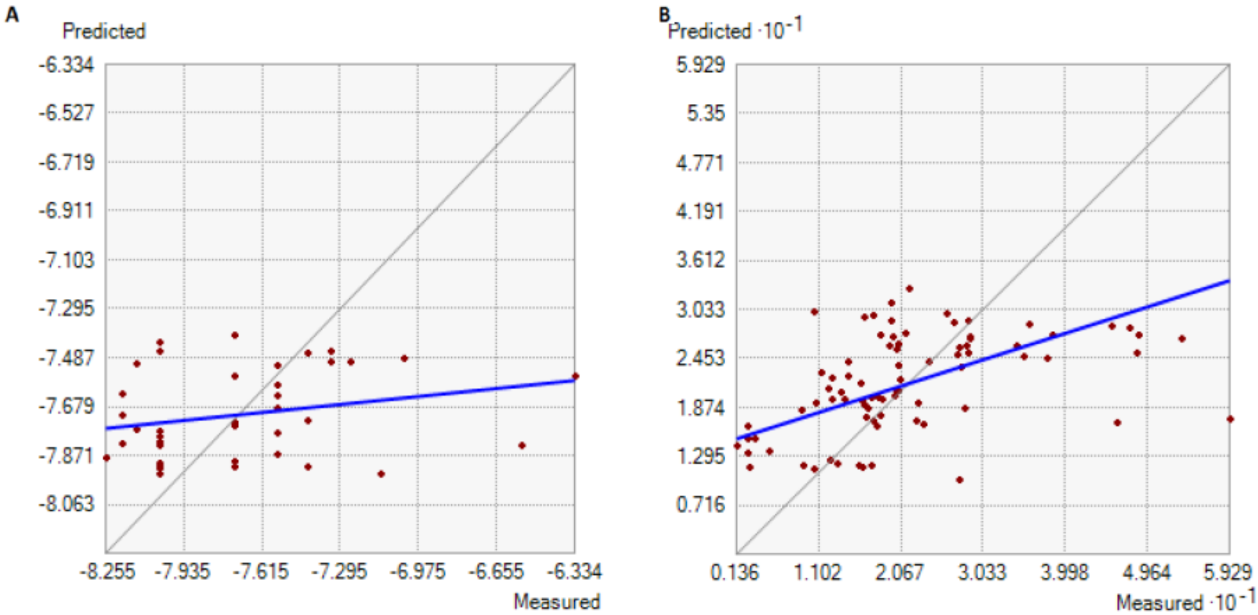


Figure SM 4.1 Comparison of K_s log-transformed values between field measurements using direct-push piezometer and laboratory measurements using constant-head upward-flow permeameter.



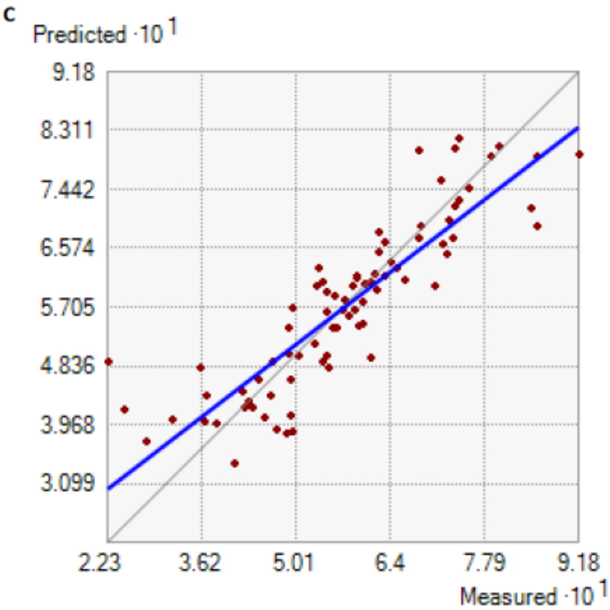


Figure SM 4.2 Plot of predicted versus measured values of (A) log-transformedKs (ms-1), (B) SOM, (C)SSM.

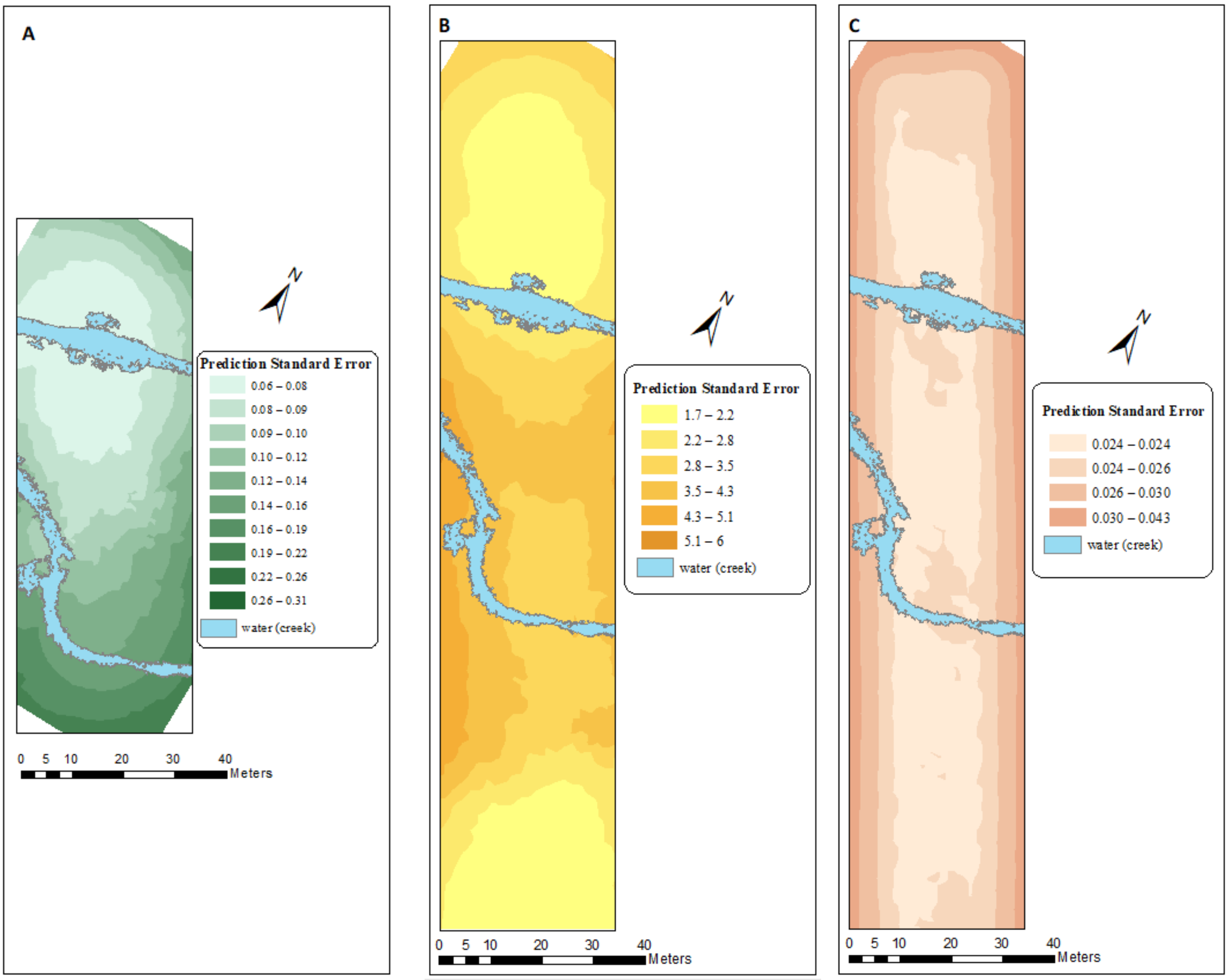


Figure SM 4.3 Contour Map of Prediction Standard Error of (A) Ks, log-transformed (m s^{-1}) (B) SOM (%w/w) (C) SSM (in metres above mean sea level)

5. Concluding Discussion

Abstract

This concluding chapter focuses on how artificial drainage of fen peatlands lead to mineralization and degradation and how rewetting can rehabilitate certain ecosystem functions. The importance of peatlands in light of the ecosystem services these deliver are revisited. Furthermore, I discuss how such management regimes may lead to changes that have implications for hydrological processes, namely water table dynamics which is a key variable which further control multiple ecosystem processes. The chapter further synthesizes how different environmental factors such as meteorological variables and microtopography act as controls on the water table and thus are important to monitor for effective peatland monitoring and restoration. The understanding of such relationships is vital, especially under the new climate regime. With increasing climate stress (eg: extreme heat and drought events), more efforts may be required for effective rewetting. Original contributions to a better understanding of peatland hydrology to sustainably restore fen peatland ecosystems and their functions are identified.

5.1. Introduction

Drainage of peatlands leads to carbon mineralization and peat degradation making carbon sinks into sources. Rewetting of drained peatlands can be an effective way for the restoration of peat ecosystems, as it results in raising the water table and thus provides anaerobic conditions required for peat accumulation. As such restoration of peatlands is a nature-based solution to mitigate climate change. Over the last decade, there has been global attention on the importance of peatlands, not only for climate regulation but also for other ecosystem services they provide. Such awareness of the array of valued ecosystem services that peatlands provide has resulted in the enactment of national and international conventions and national policies aimed at protecting peatland habitats, biodiversity, and carbon stocks. Additionally, there are initiatives in place to rehabilitate and restore peatland ecosystem functions (Page & Baird, 2016).

The United Nations Convention on Biological Diversity (CBD) and the UN Framework Convention on Climate Change (UNFCCC), adopted at the Rio Earth Summit in 1992, are the most relevant and far-reaching international conventions for peatlands. The CBD addresses the conservation of key ecosystems and protection of habitats and species and as such peatlands are prominent on the CBD lists of targets. The Ultimate objective of the UNFCCC is the stabilization of greenhouse gas (GHG) concentration, and therefore the restoration of degraded peatlands has significance for national GHG accounting. At the international level, the Ramsar Convention (1971) facilitates the conservation and wise use of all types of wetlands, and within Europe, the EU Habitats and Species Directive (1992) specifically include the mention of peatland ecosystems as priority areas for conservation and restoration where ecosystems have been changed through human interventions (Page & Baird, 2016).

As disturbed and degraded peatlands do not provide the same ecosystem services, financial resources are being spent especially in the EU for setting up appropriate restoration measures. From 1993 to 2015, the EU-LIFE nature program invested around Euros 170 Million in 80 projects, aiming to restore over 913 km^2 of peatlands in Western Europe, mostly in protected sites part of the Natura 2000 EU network. This represents less than 2% of the total remaining area of peatlands in Western Europe, most of which have been impacted by human activities to some extent (Andersen et al., 2017).

Actions for the restoration of peatlands require sound science and an improved understanding of ecosystem processes. Such process understandings also need to be ecosystem specific. Knowledge of bog hydrology for example cannot be transferred to fens, as they are quite different from each other not only in the source of water but also in their characteristic vegetation. In the same way, what works for boreal peatlands, may not be directly transferrable for temperate peatlands. Within a type of peatland, for instance, fens, there can also be substantial differences. Coastal fens may function differently than percolation fens, while the same is likely to be true for alder fens. As we increase the knowledge pool of peatland science, the relevant legislations also need to be updated. In terms of technical knowledge for peatland restoration, updating the science is particularly crucial especially with increasing vulnerability from climate change.

While the hydrology of bogs and other peatlands in the boreal region have been studied quite well, the understanding of fen hydrology in the temperate region has been vastly overlooked. For the restoration of any peatland, the water-table is a master variable, having an influence on a multitude of ecosystem structures and functions and thus the understanding of water-table dynamics is of utmost importance. Therefore, any factor which controls and/or provides feedback to water-table dynamics and related hydrological processes needs to be investigated, and their relationships need to be quantified. Field experiments have shown that fens rather than bogs, are more sensitive to warming, water table drawdown, and carbon loss (Bridgham et al., 1995; Bridgham et al., 2008; Wu & Roulet, 2014).

Therefore this doctoral dissertation has investigated the different environmental controls, such as the management regime, meteorological conditions and soil surface microtopography, on water-table dynamics and relevant soil properties.

5.2. Methodological considerations and limitations

The three **Chapters (2-4)** include methodologies that were carefully selected. Often traditional methods used for mineral soils or on soils, not within a protected area, do not work for peat soils or is not feasible. For example, in the study of heavy precipitation and the response of the water-table to precipitation events, I had first tried to apply the method called master recession curve and episodic recharge developed over the last decade with the most updated method was recently published by Nimmo and Perkins (2018). As recharge response of water-tables to rainfall events that are observed in a hydrograph is often underestimated, as the recession continues even during a rainfall event. As such the master recession curve method addresses this underestimation by constructive a master recession curve, which gives the information of where the water table would have been in the absence of rainfall. However, the application of their method was not successful for the study sites and performed very poorly. A decision was made not to use it, and a conclusion was reached that the lack of performance of the method was because (1) the method was developed for mineral soils and aquifers and not for wetlands (2) water-table of rewetted peatlands are often above the peat surface, and as such the method is not applicable. Therefore, a simpler technique was developed with the assumption that any recession of groundwater during heavy rainfall is negligible. This technique has been described in Chapter 2 (section 2.2. Methods).

The choice of the methods in chapter 3 (regarding quantifying meteorological effects on the water-table), was a difficult decision. Although there is no fixed rule regarding the minimum observations required for seasonal time-series analysis (Hyndman & Kostenko, 2007), in the absence of long-term data, it was not possible to effectively use Seasonal Autoregressive Integrated Moving Average with eXogenous variable (SARIMAX) Modelling. As such, we resorted to applying the simpler statistical method, multiple regression with months as dummy variables to control for seasonality. For understanding the underlying mechanisms of how temperature and humidity affect the water-table, we quantified evapotranspiration. There were two main options that were considered at first. One was the Penman-Monteith (see Howell & Evett, 2004), and the other was evapotranspiration determination based on diurnal groundwater fluctuation (Hays, 2003). The diurnal fluctuation method was selected, as it reflects actual ecosystem processes and includes the use of data on the water-table and calculates actual evapotranspiration rather than potential evapotranspiration, which is what the Penman-Monteith equation determines.

The method used for measuring the saturated hydraulic conductivity of the peat in the coastal fen (Chapter 4), was the only viable option which was available. The rewetted coastal fen is a protected area, which is also an important habitat for some bird species. As a result, the traditional method of collecting undisturbed soil samples extensively was not ideal. The other option of drilling boreholes all over the peatland, to conduct sludge tests for determining hydraulic conductivity was also something which is not practical. Therefore, direct-push piezometers were used to determine field hydraulic conductivity. Although the aim was to determine hydraulic conductivity at 80 points, it was possible only for 39 points, because the surface was too hard to be penetrated with the device. It must be noted that the device was originally developed to be used in bogs, not fens, and works better in soils with low compaction, and high hydraulic conductivity.

5.3. Contribution to hydrological process understanding

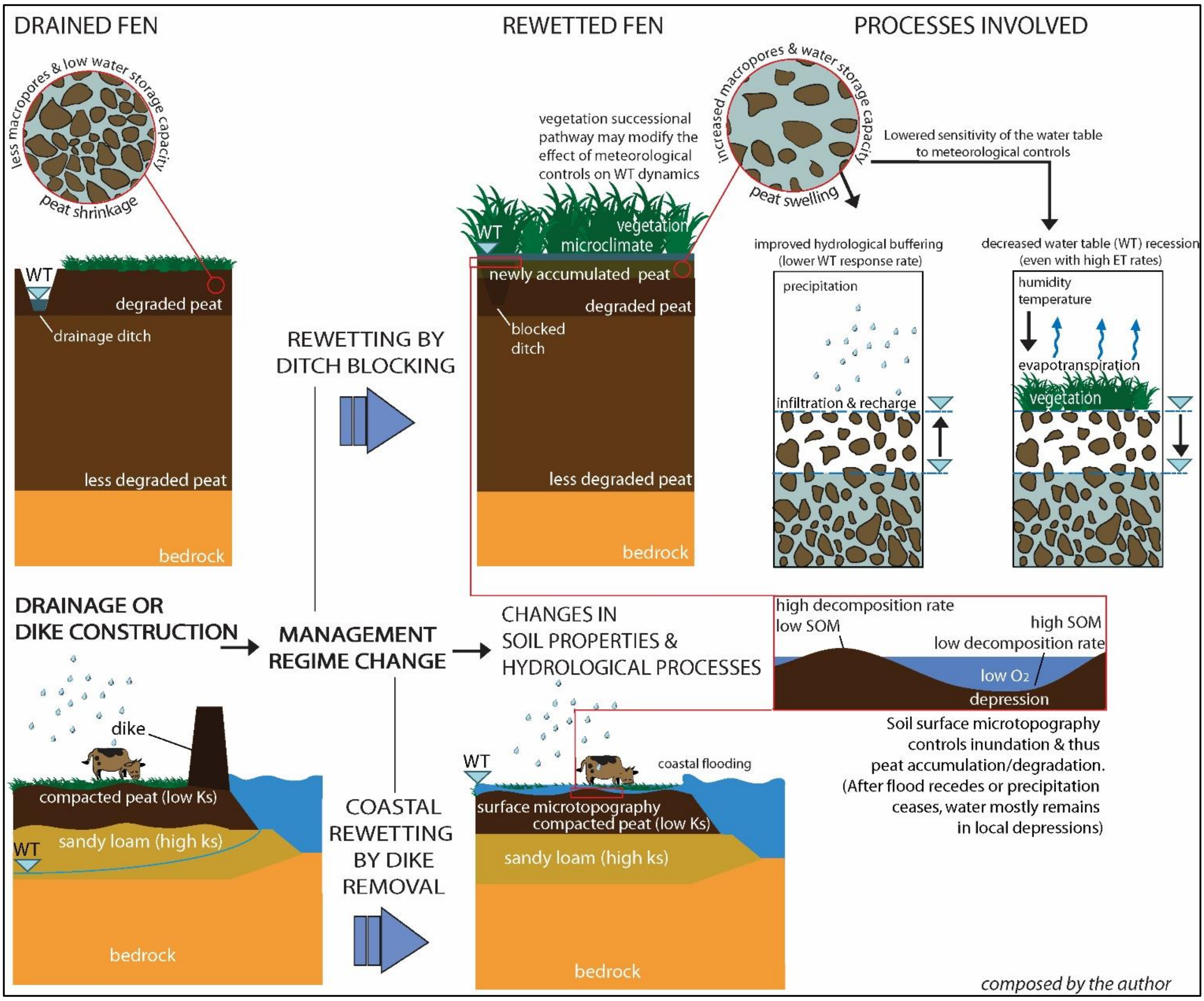


Figure 5.1 How rewetting degraded fens can alter the effect of environmental controls on hydrological processes.

As mentioned in the first chapter, hydrological processes in fen ecosystems have been largely overlooked, with bogs having received much more attention. This is particularly true for temperate fen peatlands. According to Holden (2005), "The hydrology of peatlands is fundamental to their development and decay" (p. 2892). Hydrological processes in peatlands influence gas diffusion rates, redox status, nutrient cycling and availability as well as species diversity and composition. It drives carbon sequestration and release processes and thus understanding hydrological processes in fen peatlands is vital for water resource management.

This doctoral research has contributed to a better understanding of hydrological processes in fen peatlands by investigating and quantifying (1) how precipitation drives groundwater table response and (2) quantifying the effect of temperature and humidity on water table dynamics through evapotranspiration and how such effects may vary depending on the drainage status of the fen and (3) how microtopography and soil hydrophysical properties relate to each other over space. **Figure 5.1** synthesizes the main results that were found throughout the course of this doctoral research. Drained or diked fen peatlands undergo peat degradation processes; as the peat is in contact with oxygen, it becomes oxidized and carbon mineralization process occurs. In the absence of high water levels, the peat shrinks, the surface subsides and the peat becomes more compact (high bulk density). These processes lead to lower numbers of macropores and thus lower water storage capacity. After there is a change in the water management regime, by ditch blocking or by dike removal (coastal fen) and consequent rewetting, hydrological connectivity with nearby water bodies or the sea is established. Rewetting causes the average water table to increase. A higher water table ensures anaerobic conditions which cause the decomposition rate of organic matter to decrease drastically. The vegetation in the ecosystem may undergo succession and different species composition may occur. In the presence of saturated conditions and fresh inputs of organic matter from above and below-ground biomass, new peat can accumulate, as long as the accumulation rate is higher than the decomposition rate. New peat formation has implications for how the water table physically reacts to rainfall events. The new peat will have higher macroporosity, while the higher water table can cause the previously shrunk peat to swell up. As a result the bulk density decreases and the specific yield increases. This means that it takes more water to be added to the rewetted fen peatland to have the same rise in the water-table compared to its drained pair. Thus the water table response (and the response rate) is much lower in peatlands with rewetted status.

The restoration of high water storage capacity also means that it would take a higher rate of evapotranspiration for the water table do recede a certain vertical distance, than it would in peat with very low water storage. Thus, meteorological controls on the water table is modulated by changes to the soil physical properties. A change in vegetation through ecological succession may also cause a difference in the water table dynamics. Certain species or vegetation with higher leaf area and/or biomass is likely to have higher evapotranspiration rates (Jimenez-Rodriguez et al., 2019). This has implications for the water table. However, even with high evapotranspiration, if the water storage capacity is high enough, and if there is a constant supply of water, then the water table will not be as sensitive to temperature and humidity (evapotranspiration) as it otherwise would have been (if water storage capacity was limited). In addition, the vegetation can act as a buffer between the water table and local meteorological controls such as rainfall, temperature, and humidity. Rainfall may be intercepted by vegetation and cause a difference in how much it contributes to a rise in the water table. The canopy of the vegetation can also create a microclimate that is quite different from its surroundings or above canopy weather and thus disentangle to some extent, how "external" meteorological conditions namely temperature and humidity control evapotranspiration and thus the water level. Thus, changes to vegetation following rewetting may modify how the water-table in a fen reacts to meteorological variables (different from the below-canopy microclimate).

In coastal fens, similar mechanisms are in place and in addition, flooding by the sea is what the peatland depends on for high water levels and peat accumulating conditions. In peatlands located within non-tidal coastal systems, seasonal flooding is vital. By studying the spatial structures and patterns of soil organic matter and soil surface microtopography (microelevation), it was found that soils at higher microelevation have lower organic matter content while soils located at local depressions have higher organic matter content. During coastal flooding, water enters the coastal fen and flows first to areas with lower elevation. Higher elevations are subjected to aeration, while depressions are wetter and therefore anaerobic, an ideal condition for peat formation. The same is true when it rains; the water stays longer in lower elevations and can flow from mounds to depressions. A similar effect of microtopography is also expected for non-coastal peatlands as well. For the coastal fen, a very low K_s was found for the peat horizon, while the underlying mineral horizon was determined to have saturated hydraulic conductivity of two orders of magnitude. Thus, the highly compact peat horizon may slow down the rate at which flood or rainwater infiltrates through the peat surface. While this is an interesting glimpse into understanding vertical hydrological connectivity, subsurface horizontal connectivity still remains unexplored in coastal fen peatlands.

A variable that was not within the direct scope of the doctoral work is biotic agents. How do biotic agents affect soil properties and thus hydrology? In the coastal fen, grazing by cattle (a common practice for coastal fen/salt marsh) may have an impact on biomass available for peat formation, biogeochemical processes, and compaction. Such biogenic compaction by cattle may be one explanation of why a low correlation between SOM and saturated hydraulic conductivity was found. Research in this area needs to be expanded, to get a better and holistic view of the complexity in coastal fen hydrology and development.

The collection of the research carried out as part of the doctoral thesis, was unified by the concept of "environmental controls" i.e. those environmental factors which drive other dynamics which in this case – is the water table dynamics. As mentioned earlier precipitation, air temperature, relative humidity, soil properties, and surface microtopography – along with the water management regimes - can be considered as environmental controls on the water-table. While this is the case, it must be noted that the water table itself acts as a control over other ecosystem properties and processes – thus establishing a feedback mechanism and giving rise to complexity and emergence (see Odum, 1977). For a better understanding of hydrological feedbacks in northern peatlands, Waddington et al. (2015) describe and combines several feedbacks into one conceptual model.

5.4. Implications for fen restoration and future directions

For the effective restoration and management of fen peatlands, decision-makers and practitioners need to be updated with the latest science. This is particularly true during a time when global environmental change (including climate change) is occurring at an unprecedented rate. Thus, this collection of research provides important estimates of a decrease in the water table as a response to a rise in temperature. Therefore it may be expected that with rising temperatures, more effort must be put into rewetting interventions. Therefore, immediate action is required to rewet degraded fens. Peatland rewetting projects throughout Europe must re-evaluate whether the current strategies are working, especially during dry spells and drought episodes as rapid recession of water tables make peatlands more vulnerable to carbon mineralization and peat degradation. For such evaluations, long-term monitoring of water-level and meteorological conditions are required from local levels to regional scales, and thus monitoring networks should be established and maintained. Such efforts should go beyond hydrometerological parameters, and include long-term vegetation and below-canopy microclimate monitoring. Developing relationships between above-canopy weather and below-canopy microclimate over time and over the course of vegetation development is vital for restoration and management planning.

As chapter 4 showed, peatlands can be highly heterogeneous in their soil properties and their microsurface elevation. Such variations have effects on the water-table elevation and thus all other processes which depend on high water-levels. Therefore, it is important to place several water level monitoring wells within the same peatland; specific locations can be informed by the soil surface microtopography. In other words, monitoring wells should cover both high micro-elevations as well as low depressions, to enable a better assessment of water-saturated conditions within the peatland. Current rewetting efforts may not sufficiently address the wetness required at both mounds and hollows. Furthermore, lateral hydrological connectivity between the sea and adjacent coastal fens and between adjacent rivers and inland peatlands should be better understood. This is particularly important during dry spells because in the absence of sufficient rainfall, the peatlands have to depend on inflows from adjacent water sources. Such aspects should also be considered where paludiculture is being implemented following rewetting, as paludiculture is being increasingly recognized as a sustainable and promising landuse option (Tanneberger, Schröder et al., 2020).

Existing data with high temporal resolution on peatland water levels, local weather, and soil characteristics should be compiled to enable country-wide peatland hydrological assessments especially in terms of vulnerability to climate change. Where data on soil characteristics are missing, water-level and precipitation data will help assess hydrological functioning and therefore soil hydrophysical properties such as the average specific yield, as shown in Chapters 2 and 3. While such monitoring measures are in place, more resources must be invested in extensive rewetting of drained peatlands, starting with the most degraded peatlands (cf. Liu, Wrage-Mönnig et al., 2020).

References

Adams, W. A. (1973). The effect of organic matter on the bulk and true densities of some uncultivated podzolic soils. *Journal of Soil Science*, *24*(1), 10–17.

Aerts, R., & Ludwig, F. (1997). Water-table changes and nutritional status affect trace gas emissions from laboratory columns of peatland soils. *Soil Biology and Biochemistry*, *29*(11-12), 1691–1698.

Ahmad, S., Liu, H., Beyer, F., Kløve, B., & Lennartz, B. (2020). Spatial heterogeneity of soil properties in relation to microtopography in a non-tidal rewetted coastal mire. *Mires & Peat*, *26*.

Ahmad, S., Hörmann, G., Zantout, N., & Schrautzer, J. (2020). Quantifying actual evapotranspiration in fen ecosystems: Implications of management and vegetation structure: Implications of management and vegetation structure. *Ecohydrology & Hydrobiology*, *20*(1), 381–395. https://doi.org/10.1016/j.ecohyd.2020.04.001

Ahmad, S., Liu, H., Alam, S., Günther, A., Jurasinski, G., & Lennartz, B. (2020). *Meteorological controls on water table dynamics in fen peatlands depend on management regimes (Under Review)*.

Ahmad, S., Liu, H., Günther, A., Couwenberg, J., & Lennartz, B. (2020). Long-term rewetting of degraded peatlands restores hydrological buffer function. *Science of the Total Environment*, *749*, 141571. https://doi.org/10.1016/j.scitotenv.2020.141571

Akanegbu, O. (2013). *Comparison of different methods of measuring hydraulic conductivity in drained peat soils using DRAINMOD as a verification tool* [Master Thesis, University of Oulu, Oulu, Finland]. BibTeX.

Allen, R. G., Pereira, L. S., Raes, D., & Smith, M. (1998). Crop evapotranspiration-Guidelines for computing crop water requirements-FAO Irrigation and drainage paper 56. *Fao, Rome*, *300*(9), D05109.

Altdorff, D., Bechtold, M., van der Kruk, J., Vereecken, H., & Huisman, J. A. (2016). Mapping peat layer properties with multi-coil offset electromagnetic induction and laser scanning elevation data. *Geoderma*, *261*, 178–189. https://doi.org/10.1016/j.geoderma.2015.07.015

Amoozegar, A., & Warrick, A. W. (1986). Hydraulic conductivity of saturated soils: field methods. In A. Klute (Ed.), *SSSA Book Series: Vol. 5. Methods of soil analysis Part 1: Physical and Mineralogical Methods* (2nd ed., Vol. 5, pp. 735–770). American Society of Agronomy; Soil Science Society of America.

Andersen, R., Farrell, C., Graf, M., Muller, F., Calvar, E., Frankard, P., Caporn, S., & Anderson, P. (2017). An overview of the progress and challenges of peatland restoration in Western Europe. *Restoration Ecology*, 25(2), 271–282.

Anselin, L. (2019). *GeoDa Workbook: Global Spatial Autocorrelation (2): Bivariate, Differential and EB Rate Moran Scatter Plot*. https://geodacenter.github.io/workbook/5b_global_adv/lab5b.html

Baden, W., & Eggelsmann, R. (1968). The hydrologic budget of the highbogs in the Atlantic region. In *International Peat Congress*.

Bai, J., Ouyang, H., Xiao, R., Gao, J., Gao, H., Cui, B., & Huang, L. (2010). Spatial variability of soil carbon, nitrogen, and phosphorus content and storage in an alpine wetland in the Qinghai-Tibet Plateau, China. *Australian Journal of Soil Research*, *48*(8), 730–736. https://doi.org/10.1071/SR09171

Baird, A. J. (1997). Field estimation of macropore functioning and surface hydraulic conductivity in a fen peat. *Hydrological Processes*, *11*(3), 287–295.

Baird, A. J., Surridge, B. W. J., & Money, R. P. (2004). An assessment of the piezometer method for measuring the hydraulic conductivity of a Cladium mariscus—Phragmites australis root mat in a Norfolk (UK) fen. *Hydrological Processes*, *18*(2), 275–291.

Baird, A. J., Milner, A. M., Blundell, A., Swindles, G. T., & Morris, P. J. (2016). Microform-scale variations in peatland permeability and their ecohydrological implications. *Journal of Ecology*, *104*(2), 531–544. https://doi.org/10.1111/1365-2745.12530

Baldocchi, D. D., Xu, L., & Kiang, N. (2004). How plant functional-type, weather, seasonal drought, and soil physical properties alter water and energy fluxes of an oak–grass savanna and an annual grassland. *Agricultural and Forest Meteorology*, *123*(1-2), 13–39.

Bechtold, M., Tiemeyer, B., Laggner, A., Leppelt, T., Frahm, E., & Belting, S. (2014). Large-scale regionalization of water table depth in peatlands optimized for greenhouse gas emission upscaling. *Hydrology and Earth System Sciences*, *18*(9), 3319–3339. https://doi.org/10.5194/hess-18-3319-2014

Beilman, D. W., Vitt, D. H., Bhatti, J. S., & Forest, S. (2008). Peat carbon stocks in the southern Mackenzie River Basin: Uncertainties revealed in a high-resolution case study. *Global Change Biology*, *14*(6), 1221–1232. https://doi.org/10.1111/j.1365-2486.2008.01565.x

Benavides, J. C. (2015). The effect of drainage on organic matter accumulation and plant communities of high-altitude peatlands in the Colombian tropical Andes. *Mires and Peat*, *15*.

Benscoter, B. W., Vitt, D. H., & Wieder, R. K. (2005). Association of postfire peat accumulation and microtopography in boreal bogs. *Canadian Journal of Forest Research*, *35*(9), 2188–2193. https://doi.org/10.1139/x05-115

Bernardi, A. d. C., Bettiol, G. M., Mazzuco, G. G., Esteves, S. N., Oliveira, P. P.A., & Pezzopane, J. R.M. (2017). Spatial variability of soil fertility in an integrated crop livestock forest system. *Advances in Animal Biosciences*, *8*(2), 590–593.

Bernhardt, K. G., & Koch, M. (2003). Restoration of a salt marsh system: temporal change of plant species diversity and composition. *Basic and Applied Ecology*, *4*(5), 441–451.

Bevington, J., Piragnolo, D., Teatini, P., Vellidis, G., & Morari, F. (2016). On the spatial variability of soil hydraulic properties in a Holocene coastal farmland. *Geoderma*, *262*, 294–305. https://doi.org/10.1016/j.geoderma.2015.08.025

Beyer, F., Jurasinski, G., Couwenberg, J., & Grenzdörffer, G. (2019). Multisensor data to derive peatland vegetation communities using a fixed-wing unmanned aerial vehicle. *International Journal of Remote Sensing*, 1–23.

Blodau, C., Basiliko, N., & Moore, T. R. (2004). Carbon turnover in peatland mesocosms exposed to different water table levels. *Biogeochemistry*, *67*(3), 331–351.

Boelter, D. H. (1969). Physical Properties of Peats as Related to Degree of Decomposition 1. *Soil Science Society of America Journal*, *33*(4), 606–609.

Bonn, A., Allott, T., Evans, M., Joosten, H., & Stoneman, R. (Eds.). (2016). *Ecological reviews*. *Peatland restoration and ecosystem services: Science, policy, and practice*. Cambridge University Press. https://doi.org/10.1017/CBO9781139177788

Bormann, H. (2007). Analysis of the suitability of the German soil texture classification for the regional scale application of physical based hydrological model. *Advances in Geosciences*, *11*, 7–13. https://doi.org/10.5194/adgeo-11-7-2007

Bourgault, M. A., Larocque, M., & Garneau, M. (2017). Quantification of peatland water storage capacity using the water table fluctuation method. *Hydrological Processes*, *31*(5), 1184–1195. https://doi.org/10.1002/hyp.11116

Bourgault, M. A., Larocque, M., & Garneau, M. (2019). How do hydrogeological setting and meteorological conditions influence water table depth and fluctuations in ombrotrophic peatlands? *Journal of Hydrology X*, *4*. https://doi.org/10.1016/j.hydroa.2019.100032

Brandyk, T., Szatylowicz, J., Oleszczuk, R., Gnatowski, T., Parent, L. É., & Ilnicki, P. (2002). Water-related physical attributes of organic soils. In P. Ilnicki & L. E. Parent (Eds.), *Organic soils and peat materials for sustainable agriculture* (pp. 33–66). CRC Press.

Branham, J. E., & Strack, M. (2014). Saturated hydraulic conductivity in Sphagnum-dominated peatlands: do microforms matter? *Hydrological Processes*, *28*(14), 4352–4362. https://doi.org/10.1002/hyp.10228

Bridgham, S. D., Johnston, C. A., Pastor, J., & Updegraff, K. (1995). Potential feedbacks of northern wetlands on climate change. *BioScience*, *45*(4), 262–274.

Bridgham, S. D., Pastor, J., Dewey, B., Weltzin, J. F., & Updegraff, K. (2008). Rapid carbon response of peatlands to climate change. *Ecology*, *89*(11), 3041–3048.

Bridgham, S. D., Pastor, J., Updegraff, K., Malterer, T. J., Johnson, K., Harth, C., & Chen, J. (1999). Ecosystem control over temperature and energy flux in northern peatlands. *Ecological Applications*, *9*(4), 1345–1358.

Bruland, G. L., & Richardson, C. J. (2005). Hydrologic, edaphic, and vegetative responses to microtopographic reestablishment in a restored wetland. *Restoration Ecology*, *13*(3), 515–523.

Burke, W. (1967). Principles of drainage with special reference to peat. *Irish Forestry*.

Cai, Z., & Ofterdinger, U. (2016). Analysis of groundwater-level response to rainfall and estimation of annual recharge in fractured hard rock aquifers, NW Ireland. *Journal of Hydrology*, *535*, 71–84. https://doi.org/10.1016/j.jhydrol.2016.01.066

Cambardella, C. A., Moorman, T. B., Novak, J. M., Parkin, T. B., Karlen, D. L., Turco, R. F., & Konopka, A. E. (1994). Field-scale variability of soil properties in central Iowa soils. *Soil Science Society of America Journal*, *58*(5), 1501–1511. https://doi.org/10.2136/sssaj1994.03615995005800050033x

Chattopadhyay, S., Jain, R., & Chattopadhyay, G. (2009). Estimating potential evapotranspiration from limited weather data over Gangetic West Bengal, India: a neurocomputing approach. *Meteorological Applications: A Journal of Forecasting, Practical Applications, Training Techniques and Modelling*, *16*(3), 403–411.

Childs, E. C. (1969). *An introduction to the physical basis of soil water phenomena*. John Wiley & Sons, Incorporated.

Chin, D. A. (2008). Phenomenological models of hydrologic processes in south Florida. *Journal of Hydrology*, *349*(1-2), 230–243. https://doi.org/10.1016/j.jhydrol.2007.11.003

Chmura, G. L., Anisfeld, S. C., Cahoon, D. R., & Lynch, J. C. (2003). Global carbon sequestration in tidal, saline wetland soils. *Global Biogeochemical Cycles*, *17*(4), 22-1.

Clymo, R. S. (1983). Peat. In A. J. P. Gore (Ed.), *Ecosystems of the world. Mires: swamp, bog, fen and moor.* Elsevier.

Conway, V. M., & Millar, A. (1960). The hydrology of some small peat-covered catchments in the N. Pennines. *Journal of the Institute of Water Engineers*, *14*, 415–424.

Cooper, D. J., Sanderson, J. S., Stannard, D. I., & Groeneveld, D. P. (2006). Effects of long-term water table drawdown on evapotranspiration and vegetation in an arid region phreatophyte community. *Journal of Hydrology*, *325*(1-4), 21–34.

Cooper, D. J., Sueltenfuss, J., Oyague, E., Yager, K., Slayback, D., Caballero, E. M. C., Argollo, J., & Mark, B. G. (2019). Drivers of peatland water table dynamics in the central Andes, Bolivia and Peru. *Hydrological Processes*, *33*(13), 1913–1925.

Cresto-Aleina, F., Runkle, B. R. K., Kleinen, T., Kutzbach, L., Schneider, J., & Brovkin, V. (2015). Modeling micro-topographic controls on boreal peatland hydrology and methane fluxes. *Biogeosciences*, *12*, 5689–5704.

D'Acunha, B., Lee, S. C., & Johnson, M. S. (2018). Ecohydrological responses to rewetting of a highly impacted raised bog ecosystem. *Ecohydrology*, *11*(1). https://doi.org/10.1002/eco.1922

Dargie, G. C., Lewis, S. L., Lawson, I. T., Mitchard, E. T. A., Page, S. E., Bocko, Y. E., & Ifo, S. A. (2017). Age, extent and carbon storage of the central Congo Basin peatland complex. *Nature*, *542*(7639), 86–90. https://doi.org/10.1038/nature21048

De Vos, B., Vandecasteele, B., Deckers, J., & Muys, B. (2005). Capability of loss-on-ignition as a predictor of total organic carbon in non-calcareous forest soils. *Communications in Soil Science and Plant Analysis*, *36*(19-20), 2899–2921. https://doi.org/10.1080/00103620500306080

DIN ISO (2002-08). *11277: Soil quality-Determination of particle size distribution in mineral soil material-Method by sieving and sedimentation* (Standard 11277:2002-08). Berlin. Beuth.

Dolan, T. J., Hermann, A. J., Bayley, S. E., & Zoltek Jr, J. (1984). Evapotranspiration of a Florida, USA, freshwater wetland. *Journal of Hydrology*, *74*(3-4), 355–371.

Duell, L. F. W. (1988). *Estimates of evapotranspiration in alkaline scrub and meadow communities of Owens Valley, California, using the Bowen-ratio, eddy-correlation, and Penman-combination methods* (Vol. 88). US Geological Survey.

Dunne, T., Zhang, W., & Aubry, B. F. (1991). Effects of rainfall, vegetation, and microtopography on infiltration and runoff. *Water Resources Research*, *27*(9), 2271–2285.

Eckelmann, W., Sponagel, H., Grottenthaler, W., Hartmann, K.-J., Hartwich, R., Janetzko, P., Joisten, H., Kühn, D., Sabel, K.-J., & Traidl, R. (Eds.). (2006). *Bodenkundliche Kartieranleitung. KA5*. Schweizerbart'sche Verlagsbuchhandlung.

Eurostat. *Eurostat: Your key to European statistics* [NUTS 2021 (NUTS 0)]. https://ec.europa.eu/eurostat/web/nuts/nuts-maps

Feng, X., Deventer, M. J., Lonchar, R., Ng, G. C., Sebestyen, S. D., Roman, D. T., Griffis, T. J., Millet, D. B., & Kolka, R. K. (2020). Climate sensitivity of peatland methane emissions mediated by seasonal hydrologic dynamics. *Geophysical Research Letters*, *47*(17), e2020GL088875.

Ferone, J. M., & Devito, K. J. (2004). Shallow groundwater–surface water interactions in pond–peatland complexes along a Boreal Plains topographic gradient. *Journal of Hydrology*, *292*(1-4), 75–95.

Fox, D. M., Le Bissonnais, Y., & Quétin, P. (1998). The implications of spatial variability in surface seal hydraulic resistance for infiltration in a mound and depression microtopography. *Catena*, *32*(2), 101–114. https://doi.org/10.1016/S0341-8162(98)00043-5

Gallego-Sala, A. V., Charman, D. J., Brewer, S., Page, S. E., Prentice, I. C., Friedlingstein, P., Moreton, S., Amesbury, M. J., Beilman, D. W., Bjorck, S., Blyakharchuk, T., Bochicchio, C., Booth, R. K., Bunbury, J., Camill, P., Carless, D., Chimner, R. A., Clifford, M., Cressey, E., . . . Zhao, Y. (2018). Latitudinal limits to the predicted increase of the peatland carbon sink with warming. *Nature Climate Change*, *8*(10), 907-+. https://doi.org/10.1038/s41558-018-0271-1

GeoPortal.MV. *GeoPortal.MV: GDI MV DGM5 WMS*. http://www.geodaten-mv.de/dienste/dgm_wms?

Glaser, P. H. (1999). The distribution and origin of mire pools. In V. Standen, J. H. Tallis, & R. Meade (Eds.), *Patterned Mires and Mire Pools: Origin and Development, Flora and Fauna: Proceedings, University of Durham, 6-7 April 1998* (pp. 4–25). British Ecological Society.

Glendell, M., Granger, S. J., Bol, R., & Brazier, R. E. (2014). Quantifying the spatial variability of soil physical and chemical properties in relation to mitigation of diffuse water pollution. *Geoderma*, *214-215*, 25–41. https://doi.org/10.1016/j.geoderma.2013.10.008

Goodbrand, A., Westbrook, C. J., & van der Kamp, G. (2019). Hydrological functions of a peatland in a Boreal Plains catchment. *Hydrological Processes*, *33*(4), 562–574.

Green, S. M., Baird, A. J., Holden, J., Reed, D., Birch, K., & Jones, P. (2017). An experimental study on the response of blanket bog vegetation and water tables to ditch blocking. *Wetlands Ecology and Management*, *25*(6), 703–716. https://doi.org/10.1007/s11273-017-9545-z

Grover, S. P.P., & Baldock, J. A. (2013). The link between peat hydrology and decomposition: Beyond von Post. *Journal of Hydrology*, *479*, 130–138. https://doi.org/10.1016/j.jhydrol.2012.11.049

Günther, A., Barthelmes, A., Huth, V., Joosten, H., Jurasinski, G., Koebsch, F., & Couwenberg, J. (2020). Prompt rewetting of drained peatlands reduces climate warming despite methane emissions. *Nature Communications*, *11*(1), 1644. https://doi.org/10.1038/s41467-020-15499-z

Hays, K. B. (2003). *Water use by salt cedar saltcedar (Tamarix sp.) and associated vegetation on the Canadian, Colorado, and Pecos Rivers in Texas* [Master]. Texas A&M University, Texas, USA. http://oaktrust.library.tamu.edu/handle/1969.1/ETD-TAMU-2003-THESIS-H394

Helbig, M., Waddington, J. M., Alekseychik, P., Amiro, B. D., Aurela, M., Barr, A. G., Black, T. A., Blanken, P. D., Carey, S. K., Chen, J., Chi, J., Desai, A. R., Dunn, A., Euskirchen, E. S., Flanagan, L. B., Forbrich, I., Friborg, T., Grelle, A., Harder, S., . . . Zyrianov, V. (2020). Increasing contribution of peatlands to boreal evapotranspiration in a warming climate. *Nature Climate Change*, *10*(6), 555-+. https://doi.org/10.1038/s41558-020-0763-7

Helbig, M., Waddington, J. M., Alekseychik, P., Amiro, B., Aurela, M., Barr, A. G., Black, T. A., Carey, S. K., Chen, J., & Chi, J. (2020). The biophysical climate mitigation potential of boreal peatlands during the growing season. *ENVIRONMENTAL RESEARCH LETTERS*, *15*(10), 104004.

Heliotis, F. D. (1989). Water storage capacity of wetland used for wastewater treatment. *Journal of Environmental Engineering (United States)*, *115*(4), 822–834. https://doi.org/10.1061/(ASCE)0733-9372(1989)115:4(822)

Heliotis, F. D., & DeWitt, C. B. (1987). Rapid water table responses to rainfall in a northern peatland. *JAWRA Journal of the American Water Resources Association*, *23*(6), 1011–1016. https://doi.org/10.1111/j.1752-1688.1987.tb00850.x

Heller, C., & Zeitz, J. (2012). Stability of soil organic matter in two northeastern German fen soils: The influence of site and soil development. *Journal of Soils and Sediments*, *12*(8), 1231–1240. https://doi.org/10.1007/s11368-012-0500-6

Henman, J., & Poulter, B. (2008). Inundation of freshwater peatlands by sea level rise: Uncertainty and potential carbon cycle feedbacks. *Journal of Geophysical Research: Biogeosciences*, *113*(1). https://doi.org/10.1029/2006JG000395

Holden, J. (2006). Peatland hydrology. *Developments in Earth Surface Processes*, *9*, 319–346.

Holden, J., Chapman, P. J., & Labadz, J. C. (2004). Artificial drainage of peatlands: Hydrological and hydrochemical process and wetland restoration. *Progress in Physical Geography*, *28*(1), 95–123. https://doi.org/10.1191/0309133304pp403ra

Holden, J., Evans, M. G., Burt, T. P., & Horton, M. (2006). Impact of land drainage on peatland hydrology. *Journal of Environmental Quality*, *35*(5), 1764–1778. https://doi.org/10.2134/jeq2005.0477

Holden, J., Wallage, Z. E., Lane, S. N., & McDonald, A. T. (2011). Water table dynamics in undisturbed, drained and restored blanket peat. *Journal of Hydrology*, *402*(1-2), 103–114. https://doi.org/10.1016/j.jhydrol.2011.03.010

Holden, J. (2005). Peatland hydrology and carbon release: why small-scale process matters. *Philosophical Transactions of the Royal Society a: Mathematical, Physical and Engineering Sciences*, *363*(1837), 2891–2913.

Holden, J., & Burt, T. P. (2003). Runoff production in blanket peat covered catchments. *Water Resources Research*, *39*(7). https://doi.org/10.1029/2002WR001956

Hooijer, A., Page, S., Jauhiainen, J., Lee, W. A., Lu, X. X., Idris, A., & Anshari, G. (2012). Subsidence and carbon loss in drained tropical peatlands. *Biogeosciences*, *9*(3), 1053–1071. https://doi.org/10.5194/bg-9-1053-2012

Hopple, A.M., Wilson, R. M., Kolton, M., Zalman, C. A., Chanton, J. P., Kostka, J., Hanson, P. J., Keller, J. K., & Bridgham, S. D. (2020). Massive peatland carbon banks vulnerable to rising temperatures. *Nature Communications*, *11*(1), 1–7.

Howell, T. A., & Evett, SR (2004). The Penman-Monteith Method. *Evapotranspiration: Determination of Consumptive Use in Water Rights Proceedings, Continuing Legal Education in Colorado, Inc. Denver, Colorado*.

Howie, S. A., Whitfield, P. H., Hebda, R. J., Munson, T. G., Dakin, R. A., & Jeglum, J. K. (2009). Water table and vegetation response to ditch blocking: Restoration of a raised bog in southwestern british columbia. *Canadian Water Resources Journal*, *34*(4), 381–392. https://doi.org/10.4296/cwrj3404381

Hribljan, J. A. (2012). *The effect of long-term water table manipulations on vegetation, pore water, substrate quality, and carbon cycling in a northern poor fen peatland (Doctoral Dissertation)* [, Michigan Technological University]. EndNote Tagged Import Format. https://digitalcommons.mtu.edu/cgi/viewcontent.cgi?article=1131&context=etds

Hugelius, G., Loisel, J., Chadburn, S., Jackson, R. B., Jones, M., MacDonald, G., Marushchak, M., Olefeldt, D., Packalen, M., Siewert, M. B., Treat, C., Turetsky, M., Voigt, C., & Yu, Z. (2020). Large stocks of peatland carbon and nitrogen are vulnerable to permafrost thaw. *PROCEEDINGS of the NATIONAL ACADEMY of SCIENCES of the UNITED STATES of AMERICA*, *117*(34), 20438–20446. https://doi.org/10.1073/pnas.1916387117

Humpenoeder, F., Karstens, K., Lotze-Campen, H., Leifeld, J., Menichetti, L., Barthelmes, A., & Popp, A. (2020). Peatland protection and restoration are key for climate change mitigation. *ENVIRONMENTAL RESEARCH LETTERS*, *15*(10). https://doi.org/10.1088/1748-9326/abae2a

Hur, S.-O., Jung, K.-H., Sonn, Y.-K., Ha, S.-K., & Kim, J.-G. (2009). Determination of pedo-transfer function using the relation between soil particle distribution, organic matter and water movement in soil originated from limestone. *Korean Journal of Soil Science and Fertilizer*, *42*(2), 132–138.

Huryna, H., Brom, J., & Pokorny, J. (2014). The importance of wetlands in the energy balance of an agricultural landscape. *Wetlands Ecology and Management*, *22*(4), 363–381.

Hvorslev, M. J. (1951). Time lag and soil permeability in ground-water observations, Waterways Experiment Station, Corps of Engineers. *US Army, Bulletin*, *36*, 49.

Hylleberg, S., Jørgensen, C., & Sørensen, N. K. (1993). Seasonality in macroeconomic time series. *Empirical Economics*, *18*(2), 321–335.

Hyndman, R. J., Athanasopoulos, G., Bergmeir, C., Caceres, G., Chhay, L., O'Hara-Wild, M., Petropoulos, F., & Razbash, S. (2020). *Package 'forecast'*. https://cran.r-project.org/web/packages/forecast/forecast.pdf

Hyndman, R. J., & Kostenko, A. V. (2007). Minimum sample size requirements for seasonal forecasting models. *Foresight*, 6(Spring), 12–15.

Ibenthal, M. (2019). *Marine and Terrestrial Influence on Submarine Groundwater Discharge in Coastal Waters Connected to a Peatland* [Doctoral Thesis]. Georg-August-Universität Göttingen, Germany.

Intergovernmental Panel on Climate Change. (2019). *IPCC special report on climate change, desertification, land degradation, sustainable land management, food security, and greenhouse gas fluxes in terrestrial ecosystems: Summary for policymakers.*

Ivanov, K. E. (1981). *Water movement in mirelands*. Academic Press Inc.(London) Ltd.

Jaenicke, J., Rieley, J. O., Mott, C., Kimman, P., & Siegert, F. (2008). Determination of the amount of carbon stored in Indonesian peatlands. *Geoderma*, *147*(3-4), 151–158. https://doi.org/10.1016/j.geoderma.2008.08.008

Janssen, M., Böttcher, M. E., Brede, M., Burchardt, H., Forster, S., Karsten, U., Leinweber, P., Lennartz, B., Rehder, G., Schubert, H., Schulz-Vogt, H., Sokolova, I. M., Voss, M., & Jurasinski, G. (July 2019). *The Baltic TRANSCOAST approach-investigating shallow coasts as terrestrial-marine interface of water and matter fluxes.* https://doi.org/10.31223/osf.io/e7cj2

Jimenez-Rodriguez, D. C., Esquivel-Vargas, C., Coenders-Gerrits, M., & Sasa-Marin, M. (2019). Quantification of the Evaporation Rates from Six Types of Wetland Cover in Palo Verde National Park, Costa Rica. *Water*, *11*(4). https://doi.org/10.3390/w11040674

Joosten, H., & Couwenberg, J. (2001). Bilanzen zum Moorverlust: Das Beispiel Europa [Balance sheet for loss of mires: The example of Europe]. In M. Succow & H. Joosten (Eds.), *Landschaftsökologische Moorkunde [Landscape Ecology of Mires]* (2nd ed., Vol. 2, pp. 406–409). Schweizerbart Science Publishers.

Joosten, H. (2010). *The global peatland CO_2 picture: Peatland status and drainage related emissions in all countries of the world*. Wetlands International.

Joosten, H., & Clarke, D. (2002). *Wise use of mires and peatlands*. International Mire Conservation Group; International Peat Society.

Joosten, H., & Klerk, P. D. (2007). DAMOCLES: a DAshing MOnolith Cutter for fine sectioning of peats and sediments into LargE Slices. *Boreas*, *36*(1), 76–81. https://doi.org/10.1111/j.1502-3885.2007.tb01182.x

Joosten, H., Sirin, A., Couwenberg, J., Laine, J., & Smith, P. (2016). The role of peatlands in climate regulation. In A. Bonn, T. Allott, M. Evans, H. Joosten, & R. Stoneman (Eds.), *Ecological reviews. Peatland restoration and ecosystem services: Science, policy, and practice* (pp. 63–76). Cambridge University Press. https://doi.org/10.1017/CBO9781139177788.005

Joosten, H., Tanneberger, F., & Moen, A. (Eds.). (2017). *Mires and peatlands of Europe*. Schweizerbart'sche Verlagsbuchhandlung.

Jurasinski, G., Ahmad, S., Anadon-Rosell, A., Berendt, J., Beyer, F., Bill, R., Blume-Werry, G., Couwenberg, J., Günther, A., Joosten, H., Köbsch, F., Köhn, D., Koldrack, N., Kreyling, J., Leinweber, P., Lennartz, B., Liu, H., Michaelis, D., Mrotzek, A., . . . Wrage-Mönnig, N. (2020). From Understanding to Sustainable Use of Peatlands:: The WETSCAPES Approach. *Soil Systems*, *4*(1), Article 14. https://doi.org/10.3390/soilsystems4010014

Jurasinski, G., Janssen, M., Voss, M., Böttcher, M. E., Brede, M., Burchard, H., Forster, S., Gosch, L., Gräwe, U., Gründling-Pfaff, S., Haider, F., Ibenthal, M., Karow, N., Karsten, U., Kreuzburg, M., Lange, X., Leinweber, P., Massmann, G., Ptak, T., . . . Lennartz, B. (2018). Understanding the coastal ecocline: Assessing sea-land interactions at non-tidal, low-lying coasts through interdisciplinary research. *Frontiers in Marine Science*, *5*(SEP). https://doi.org/10.3389/fmars.2018.00342

Kechavarzi, C., Dawson, Q., & Leeds-Harrison, P. B. (2010). Physical properties of low-lying agricultural peat soils in England. *Geoderma*, *154*(3-4), 196–202. https://doi.org/10.1016/j.geoderma.2009.08.018

Kelly, T. J., Lawson, I. T., & Cole, L. E. S. (2017). Peat. In W. M. White (Ed.), *Encyclopedia of Geochemistry: A Comprehensive Reference Source on the Chemistry of the Earth* (pp. 1–4). Springer International Publishing. https://doi.org/10.1007/978-3-319-39193-9_187-1

Kennedy, G. W., & Price, J. S. (2004). Simulating soil water dynamics in a cutover bog. *Water Resources Research*, *40*(12), 1–13. https://doi.org/10.1029/2004WR003099

Kettridge, N., Turetsky, M. R., Sherwood, J. H., Thompson, D. K., Miller, C. A., Benscoter, B. W., Flannigan, M. D., Wotton, B. M., & Waddington, J. M. (2015). Moderate drop in water table increases peatland vulnerability to post-fire regime shift. *Scientific Reports*, *5*(1), 1–4.

Kettunen, A., Kaitala, V., Lehtinen, A., Lohila, A., Alm, J., Silvola, J., & Martikainen, P. J. (1999). Methane production and oxidation potentials in relation to water table fluctuations in two boreal mires. *Soil Biology and Biochemistry*, *31*(12), 1741–1749.

Kimmel, K., & Mander, Ü. (2010). Ecosystem services of peatlands: Implications for restoration. *Progress in Physical Geography*, *34*(4), 491–514.

Kirschbaum, M. U. F., & McMillan, A. M. S. (2018). Warming and Elevated CO 2 Have Opposing Influences on Transpiration. Which is more Important? *Current Forestry Reports*, *4*(2), 51–71.

Kluge, B., Wessolek, G., Facklam, M., Lorenz, M., & Schwärzel, K. (2008). Long-term carbon loss and CO2-C release of drained peatland soils in northeast Germany. *European Journal of Soil Science*, *59*(6), 1076–1086.

Koch, M., & Jurasinski, G. (2015). Four decades of vegetation development in a percolation mire complex following intensive drainage and abandonment. *Plant Ecology & Diversity*, *8*(1), 49–60.

Kotowski, W., Acreman, M., Grootjans, A., Klimkowska, A., Rößling, H., & Wheeler, B. (2016). Restoration of temperate fens: matching strategies with site potential. In A. Bonn, T. Allott, M. Evans, H. Joosten, & R. Stoneman (Eds.), *Ecological reviews. Peatland restoration and ecosystem services: Science, policy, and practice* (pp. 170–191). Cambridge University Press. https://doi.org/10.1017/CBO9781139177788.011

Lamers, L. P. M., Vile, M. A., Grootjans, A. P., Acreman, M. C., van Diggelen, R., Evans, M. G., Richardson, C. J., Rochefort, L., Kooijman, A. M., Roelofs, J. G. M., & Smolders, A. J. P. (2015). Ecological restoration of rich fens in Europe and North America: From trial and error to an evidence-based approach. *Biological Reviews of the Cambridge Philosophical Society*, *90*(1), 182–203. https://doi.org/10.1111/brv.12102

Lampe, R., & Wolrab, B. (1996). Geländeklimatologische und hydrologische Untersuchungen. *Natur Und Naturschutz in Mecklenburg-Vorpommern*, *32*, 43–55.

Lautz, L. K. (2008). Estimating groundwater evapotranspiration rates using diurnal water-table fluctuations in a semi-arid riparian zone. *Hydrogeology Journal*, *16*(3), 483–497. https://doi.org/10.1007/s10040-007-0239-0

Lavoie, M., Paré, D., Fenton, N., Groot, A., & Taylor, K. (2005). Paludification and management of forested peatlands in Canada: a literature review. *Environmental Reviews*, *13*(2), 21–50. https://doi.org/10.1139/a05-006

Lee, S.-I. (2017). Correlation and Spatial Autocorrelation. In Shekhar, Shashi and Xiong, Hui and Zhou, Xun (Ed.), *Encyclopedia of GIS* (Vol. 1, pp. 360–368). Springer International Publishing.

Lefèvre, C., Rekik, F., Alcantara, V., Wiese, L., & others. (2017). *Soil organic carbon: the hidden potential*. Food and Agriculture Organization of the United Nations (FAO).

Leifeld, J., & Menichetti, L. (2018). The underappreciated potential of peatlands in global climate change mitigation strategies. *Nature Communications*, *9*(1), Article 1071, 1–7. https://doi.org/10.1038/s41467-018-03406-6

Leifeld, J., Wüst-Galley, C., & Page, S. (2019). Intact and managed peatland soils as a source and sink of GHGs from 1850 to 2100. *Nature Climate Change*, *9*(12), 945–947. https://doi.org/10.1038/s41558-019-0615-5

Lembrechts, J. J., Aalto, J., Ashcroft, M. B., Frenne, P. de, Kopecký, M., Lenoir, J., Luoto, M., Maclean, I. M. D., Roupsard, O., & Fuentes-Lillo, E. (2020). SoilTemp: a global database of near-surface temperature. *Global Change Biology*, 6616–6629.

Lennartz, B., & Liu, H. (2019). Hydraulic Functions of Peat Soils and Ecosystem Service. *Frontiers in Environmental Science*, *7*, Article 92. https://doi.org/10.3389/fenvs.2019.00092

Levison, J., Larocque, M., Fournier, V., Gagné, S., Pellerin, S., & Ouellet, M. A. (2014). Dynamics of a headwater system and peatland under current conditions and with climate change. *Hydrological Processes*, *28*(17), 4808–4822.

Lewis, C., Albertson, J., Xu, X., & Kiely, G. (2012). Spatial variability of hydraulic conductivity and bulk density along a blanket peatland hillslope. *Hydrological Processes*, *26*(10), 1527–1537. https://doi.org/10.1002/hyp.8252

Lindsay, R. (2016). Peatland (Mire Types): Based on Origin and Behavior of Water, Peat Genesis, Landscape Position, and Climate. In *The Wetland Book.* Springer. https://doi.org/10.1007/978-94-007-6173-5_279-1

Lindsay, R., & Andersen, R. (2016). Peat. In Finlayson, C. Max and Milton, G. Randy and Prentice, R. Crawford and Davidson, Nick C (Ed.), *The Wetland Book: II: Distribution, Description and Conservation* (pp. 1–6). Springer Netherlands. https://doi.org/10.1007/978-94-007-6173-5_274-1

Liu, H., Janssen, M., & Lennartz, B. (2016). Changes in flow and transport patterns in fen peat following soil degradation. *European Journal of Soil Science*, *67*(6), 763–772. https://doi.org/10.1111/ejss.12380

Liu, H., Zak, D., Rezanezhad, F., & Lennartz, B. (2019). Soil degradation determines release of nitrous oxide and dissolved organic carbon from peatlands. *ENVIRONMENTAL RESEARCH LETTERS*, *14*(9). https://doi.org/10.1088/1748-9326/ab3947

Liu, H., & Lennartz, B. (2019a). Hydraulic properties of peat soils along a bulk density gradient—A meta study. *Hydrological Processes*, *33*(1), 101–114. https://doi.org/10.1002/hyp.13314

Liu, H., & Lennartz, B. (2019b). Short Term Effects of Salinization on Compound Release from Drained and Restored Coastal Wetlands. *Water*, *11*(8), 1549.

Liu, H., Price, J., Rezanezhad, F., & Lennartz, B. (2020). Centennial-Scale Shifts in Hydrophysical Properties of Peat Induced by Drainage. *Water Resources Research*, *56*(10), 141. https://doi.org/10.1029/2020WR027538

Liu, H., Wrage-Mönnig, N., & Lennartz, B. (2020). Rewetting strategies to reduce nitrous oxide emissions from European peatlands. *Communications Earth & Environment*, *1*(1), 1–7. https://doi.org/10.1038/s43247-020-00017-2

Ma, Y., Minasny, B., Malone, B. P., & Mcbratney, A. B. (2019). Pedology and digital soil mapping (DSM). *European Journal of Soil Science*, *70*(2), 216–235. https://doi.org/10.1111/ejss.12790

Maki, J. E., Hoffman, D. M., & Berk, R. A. (1978). A time series analysis of the impact of a water conservation campaign. *Evaluation Quarterly*, *2*(1), 107–118.

Malhotra, A., Roulet, N. T., Wilson, P., Giroux-Bougard, X., & Harris, L. I. (2016). Ecohydrological feedbacks in peatlands: an empirical test of the relationship among vegetation, microtopography and water table. *Ecohydrology*, *9*(7), 1346–1357.

Malloy, S., & Price, J. S. (2014). Fen restoration on a bog harvested down to sedge peat: A hydrological assessment. *Ecological Engineering*, *64*, 151–160.

Maltby, E. (1997). Peatlands:: The science case for conservation and sound management. In L. Parkyn, R. E. Stoneman, & H. A. P. Ingram (Eds.), *Conserving peatlands* (pp. 121–131). CAB International.

Maréchal, J. C., Dewandel, B., Ahmed, S., Galeazzi, L., & Zaidi, F. K. (2006). Combined estimation of specific yield and natural recharge in a semi-arid groundwater basin with irrigated agriculture. *Journal of Hydrology*, *329*(1-2), 281–293. https://doi.org/10.1016/j.jhydrol.2006.02.022

Mazur, M. L. C., Wiley, M. J., & Wilcox, Douglas, A (2014). Estimating evapotranspiration and groundwater flow from water-table fluctuations for a general wetland scenario. *Ecohydrology*, *7*(2), 378–390. https://doi.org/10.1002/eco.1356

McCarter, C. P.R., Rezanezhad, F., Quinton, W. L., Gharedaghloo, B., Lennartz, B., Price, J., Connon, R., & van Cappellen, P. (2020). Pore-scale controls on hydrological and geochemical processes in peat: Implications on interacting processes. *Earth-Science Reviews*, 103227.

Menberu, M. W., Tahvanainen, T., Marttila, H., Irannezhad, M., Ronkanen, A. K., Penttinen, J., & Kløve, B. (2016). Water-table-dependent hydrological changes following peatland forestry drainage and restoration: Analysis of restoration success. *Water Resources Research*, *52*(5), 3742–3760. https://doi.org/10.1002/2015WR018578

Meyboom, P. (1966). Unsteady groundwater flow near a willow ring in hummocky moraine. *Journal of Hydrology*, *4*, 38–62.

Michaelis, D., Mrotzek, A., & Couwenberg, J. (2020). Roots, Tissues, Cells and Fragments—How to Characterize Peat from Drained and Rewetted Fens. *Soil Systems*, *4*(1), 12. https://doi.org/10.3390/soilsystems4010012

Minasny, B., Berglund, Ö., Connolly, J., Hedley, C., de Vries, F., Gimona, A., Kempen, B., Kidd, D., Lilja, H., Malone, B., McBratney, A., Roudier, P., O'Rourke, S., Rudiyanto, Padarian, J., Poggio, L., ten Caten, A., Thompson, D., Tuve, C., & Widyatmanti, W. (2019). Digital mapping of peatlands – A critical review. *Earth-Science Reviews*, *196*. https://doi.org/10.1016/j.earscirev.2019.05.014

Moore, P. D. (1989). The ecology of peat-forming processes: a review. *INTERNATIONAL JOURNAL of COAL GEOLOGY*, *12*(1-4), 89–103.

Morris, P. J., Baird, A. J., Eades, P. A., & Surridge, B. W.J. (2019). Controls on Near-Surface Hydraulic Conductivity in a Raised Bog. *Water Resources Research.* Advance online publication. https://doi.org/10.1029/2018WR024566

Morris, P. J., Waddington, J. M., Benscoter, B. W., & Turetsky, M. R. (2011). Conceptual frameworks in peatland ecohydrology: looking beyond the two-layered (acrotelm-catotelm) model. *Ecohydrology*, *4*(1), 1–11. https://doi.org/10.1002/eco.191

Mould, D. J., Frahm, E., Salzmann, T., Miegel, K., & Acreman, M. C. (2010). Evaluating the use of diurnal groundwater fluctuations for estimating evapotranspiration in wetland environments: case studies in southeast England and northeast Germany. *Ecohydrology*, *3*(3), 294–305. https://doi.org/10.1002/eco.108

Mrotzek, A., Michaelis, D., Günther, A., Wrage-Mönnig, N., & Couwenberg, J. (2020). Mass Balances of a Drained and a Rewetted Peatland: on Former Losses and Recent Gains. *Soil Systems*, *4*(1), 16. https://doi.org/10.3390/soilsystems4010016

Mustamo, P., Hyvarinen, M., Ronkanen, A.-K., & Klove, B. (2016). Physical properties of peat soils under different land use options. *Soil Use and Management*, *32*(3), 400–410. https://doi.org/10.1111/sum.12272

Nath, T. N., & Krishna, B. G. (2014). Influence of soil texture and total organic matter content on soil hydraulic conductivity of some selected Tea growing soils in Dibrugarh district of Assam, India. *Journal of International Research*, 2–9.

Nichols, W. D. (1994). Groundwater discharge by phreatophyte shrubs in the Great Basin as related to depth to groundwater. *Water Resources Research*, *30*(12), 3265–3274.

Niedermeier, A., & Robinson, J. S. (2007). Hydrological controls on soil redox dynamics in a peat-based, restored wetland. *Geoderma*, *137*(3-4), 318–326. https://doi.org/10.1016/j.geoderma.2006.08.027

Nijp, J. J., Limpens, J., Metselaar, K., Peichl, M., Nilsson, M. B., van der Zee, S. E. A. T. M., & Berendse, F. (2015). Rain events decrease boreal peatland net CO2 uptake through reduced light availability. *Global Change Biology*, *21*(6), 2309–2320. https://doi.org/10.1111/gcb.12864

Nimmo, J. R., & Perkins, K. S. (2018). Episodic master recession evaluation of groundwater and streamflow hydrographs for water-resource estimation. *Vadose Zone Journal*, *17*(1), 1–25.

Nugent, K. A., Strachan, I. B., Roulet, N. T., Strack, M., Frolking, S., & Helbig, M. (2019). Prompt active restoration of peatlands substantially reduces climate impact. *ENVIRONMENTAL RESEARCH LETTERS*, *14*(12). https://doi.org/10.1088/1748-9326/ab56e6

Odum, E. P. (1977). The emergence of ecology as a new integrative discipline. *Science*, *195*(4284), 1289–1293.

Oleszczuk, R., & Brandyk, T. (2008). The analysis of shrinkage-swelling behavior of peat-moorsh soil aggregates during drying-wetting cycles. *Agronomy Research*, *6*(1), 131–140.

Öquist, M., & Sundh, I. (1998). Effects of a transient oxic period on mineralization of organic matter to CH4 and CO2 in anoxic peat incubations. *Geomicrobiology Journal*, *15*(4), 325–333.

Page, S. E., & Baird, A. J. (2016). Peatlands and global change: response and resilience. *Annual Review of Environment and Resources*, *41*.

Page, S. E., Rieley, J. O., & Banks, C. J. (2011). Global and regional importance of the tropical peatland carbon pool. *Global Change Biology*, *17*(2), 798–818. https://doi.org/10.1111/j.1365-2486.2010.02279.x

Paz Ferreiro, J., Pereira De Almeida, V., Cristina Alves, M., Aparecida De Abreu, C., Vieira, S. R., & Vidal Vázquez, E. (2016). Spatial Variability of Soil Organic Matter and Cation Exchange Capacity in an Oxisol under Different Land Uses. *Communications in Soil Science and Plant Analysis*, *47*, 75–89. https://doi.org/10.1080/00103624.2016.1232099

Perie, C., & Ouimet, R. (2008). Organic carbon, organic matter and bulk density relationships in boreal forest soils. *Canadian Journal of Soil Science*, *88*(3), 315–325.

Petrone, R. M., Price, J. S., Waddington, J. M., & Waldow, H. v. (2003). Effects of a changing surface cover on the moisture and energy exchange of a restored peatland. *J. Hydrol.*

Philippov, D. A., & Yurchenko, V. V. (2019). Data on air temperature, relative humidity and dew point in a boreal Sphagnum bog and an upland site (Shichengskoe mire system, North-Western Russia). *Data in Brief*, *25*, 104156.

Postila, H., Ronkanen, A.-K., Marttila, H., & Kløve, B. (2015). Hydrology and hydraulics of treatment wetlands constructed on drained peatlands. *Ecological Engineering*, *75*, 232–241.

Price, J. S. (1996). Hydrology and microclimate of a partly restored cutover bog, Quebec. *Hydrological Processes*, *10*(10), 1263–1272.

Price, J. S. (2003). Role and character of seasonal peat soil deformation on the hydrology of undisturbed and cutover peatlands. *Water Resources Research*, *39*(9). https://doi.org/10.1029/2002WR001302

Pronger, J., Schipper, L. A., Hill, R. B., Campbell, D. I., & McLeod, M. (2014). Subsidence Rates of Drained Agricultural Peatlands in New Zealand and the Relationship with Time since Drainage. *Journal of Environmental Quality*, *43*(4), 1442–1449. https://doi.org/10.2134/jeq2013.12.0505

Qiu, C., Zhu, D., Ciais, P., Guenet, B., & Peng, S. (2020). The role of northern peatlands in the global carbon cycle for the 21st century. *Global Ecology and Biogeography*, *29*(5), 956–973.

R: A Language and Environment for Statistical Computing [Computer software]. (2019). Vienna, Austria. https://www.R-project.org/

Ramirez, B. V.H., & Harmsen, W. E. (2011). Water vapor flux in agroecosystems: methods and models review. In L. Labedzki (Ed.), *Evapotranspiration* (pp. 3–48). InTech.

Rawls, W. J. (2004). Effect of soil organic carbon on soil hydraulic properties. In Y. Pachepsky & W. J. Rawls (Eds.), *Development of pedotransfer functions in soil hydrology* (Vol. 30, pp. 95–114). Elsevier.

Rezanezhad, F., Price, J. S., Quinton, W. L., Lennartz, B., Milojevic, T., & Van Cappellen, P. (2016). Structure of peat soils and implications for water storage, flow and solute transport: A review update for geochemists. *Chemical Geology*, *429*, 75–84. https://doi.org/10.1016/j.chemgeo.2016.03.010

Robinson, M. (1980). The effect of pre-afforestation drainage on the stream-flow and water quality of a small upland catchment.

Robinson, M. (1985). Hydrological effects of moorland gripping: a re-appraisal of the Moor House Research. *J. Environ. Manage.;(United States)*, *21*(3).

Rogers, K., Kelleway, J. J., Saintilan, N., Megonigal, J. P., Adams, J. B., Holmquist, J. R., Lu, M., Schile-Beers, L., Zawadzki, A., Mazumder, D., & Woodroffe, C. D. (2019). Wetland carbon storage controlled by millennial-scale variation in relative sea-level rise. *Nature*, 567(7746), 91–95. https://doi.org/10.1038/s41586-019-0951-7

Romanov, V. V. (1968). *Hydrophysics of Bogs:(Gidrofizika Bolot)*. Israel Program for Scientific Translations [available from the US Department …

Ronkanen, A.-K., Kløve, B., & others (2005). Hydraulic soil properties of peatlands treating municipal wastewater and peat harvesting runoff. *Suo*, *56*(2), 43–56.

Rudiyanto, Minasny, B., Setiawan, B. I., Saptomo, S. K., & Mcbratney, A. B. (2018). Open digital mapping as a cost-effective method for mapping peat thickness and assessing the carbon stock of tropical peatlands. *Geoderma*, *313*, 25–40. https://doi.org/10.1016/j.geoderma.2017.10.018

Ruseckas, J., & Grigaliūnas, V. (2008). Effect of drain-blocking and meteorological factors on ground water table fluctuations in kamanos mire. *Journal of Environmental Engineering and Landscape Management*, *16*(4), 168–177.

Rydin, H., & Jeglum, J. K. (2006). *The biology of peatlands*. *The biology of habitats*. Oxford University.

Rydin, H., & Jeglum, J. K. (2013). *The biology of peatlands* (Second edition). *The biology of habitats series*. Oxford University Press.

Saarinen, T., Mohammadighavam, S., Marttila, H., & Klove, B. (2013). Impact of peatland forestry on runoff water quality in areas with sulphide-bearing sediments; how to prevent acid surges. *Forest Ecology and Management*, *293*, 17–28. https://doi.org/10.1016/j.foreco.2012.12.029

Säurich, A., Tiemeyer, B., Dettmann, U., & Don, A. (2019). How do sand addition, soil moisture and nutrient status influence greenhouse gas fluxes from drained organic soils? *Soil Biology and Biochemistry*, *135*, 71–84. https://doi.org/10.1016/j.soilbio.2019.04.013

Schaap, M. G., Leij, F. J., & van Genuchten, M. T. (2001). ROSETTA: a computer program for estimating soil hydraulic parameters with hierarchical pedotransfer functions. *Journal of Hydrology*, *251*(3-4), 163–176.

Scharlemann, J. P.W., Tanner, E. V.J., Hiederer, R., & Kapos, V. (2014). Global soil carbon: Understanding and managing the largest terrestrial carbon pool. *Carbon Management*, *5*(1), 81–91. https://doi.org/10.4155/cmt.13.77

Schrautzer, J., Rinker, A., Jensen, K., Müller, F., Schwartze, P., & Dierßen, K. (2007). Succession and restoration of drained fens: perspectives from northwestern Europe. In L. R. Walker, J. Walker, & R. J. Hobbs (Eds.), *Springer Series on Environmental Management. Linking restoration and ecological succession* (pp. 90–120). Springer.

Schrautzer, J., Sival, F., Breuer, M., Runhaar, H., & Fichtner, A. (2013). Characterizing and evaluating successional pathways of fen degradation and restoration. *Ecological Indicators*, *25*, 108–120. https://doi.org/10.1016/j.ecolind.2012.08.018

Schulte, M. L., McLaughlin, D. L., Wurster, F. C., Balentine, K., Speiran, G. K., Aust, W. M., Stewart, R. D., Varner, J. M., & Jones, C. N. (2019). Linking ecosystem function and hydrologic regime to inform restoration of a forested peatland. *Journal of Environmental Management*, *233*, 342–351.

Schwieger, S., Kreyling, J., Couwenberg, J., Smiljanić, M., Weigel, R., Wilmking, M., & Blume-Werry, G. (2020). Wetter is Better: Rewetting of Minerotrophic Peatlands Increases Plant Production and Moves Them Towards Carbon Sinks in a Dry Year. *Ecosystems*, 1–17.

Shaw, E. M., Beven, K. J., Chappell, N. A., & Lamb, R. (2010). *Hydrology in practice*. CRC Press.

Shi, X., Thornton, P. E., Ricciuto, D. M., Hanson, P. J., Mao, J., Sebestyen, S. D., Griffiths, N. A., & Bisht, G. (2015). Representing northern peatland microtopography and hydrology within the Community Land Model. *Biogeosciences Discussions (Online)*, *12*(4).

Sofi, M. S., Bhat, S. U., Rashid, I., & Kuniyal, J. C. (2020). The natural flow regime: A master variable for maintaining river ecosystem health. *Ecohydrology.* Advance online publication. https://doi.org/10.1002/eco.2247

Sridharan, S., Tunstall, H., Lawder, R., & Mitchell, R. (2007). An exploratory spatial data analysis approach to understanding the relationship between deprivation and mortality in Scotland. *Social Science & Medicine*, 65(9), 1942–1952. https://doi.org/10.1016/j.socscimed.2007.05.052

Strack, M., Waddington, J. M., Bourbonniere, R. A., Buckton, E. L., Shaw, K., Whittington, P., & Price, J. S. (2008). Effect of water table drawdown on peatland dissolved organic carbon export and dynamics. *Hydrological Processes*, *22*(17), 3373–3385.

Succow, M., & Joosten, H. (Eds.). (2001). *Landschaftsökologische Moorkunde [Landscape Ecology of Mires]* (2nd). Schweizerbart Science Publishers. http://www.schweizerbart.de//publications/detail/isbn/9783510651986/Landschaftsokologische_Moorkunde_Hrsg

Swindles, G. T., Morris, P. J., Mullan, D. J., Payne, R. J., Roland, T. P., Amesbury, M. J., Lamentowicz, M., Turner, T. E., Gallego-Sala, A., & Sim, T. (2019). Widespread drying of European peatlands in recent centuries. *Nature Geoscience*, *12*(11), 922–928.

Tanneberger, F., Appulo, L., Ewert, S., Lakner, S., Brolchain, N. O., Peters, J., & Wichtmann, W. (2020). The Power of Nature-Based Solutions: How Peatlands Can Help Us to Achieve Key EU Sustainability Objectives. *ADVANCED SUSTAINABLE SYSTEMS.* Advance online publication. https://doi.org/10.1002/adsu.202000146

Tanneberger, F., Schröder, C., Hohlbein, M., Lenschow, U., Permien, T., Wichmann, S., & Wichtmann, W. (2020). Climate Change Mitigation through Land Use on Rewetted Peatlands–Cross-Sectoral Spatial Planning for Paludiculture in Northeast Germany. *Wetlands*, 1–12.

Tarnocai, C. (2009). The impact of climate change on Canadian peatlands. *Canadian Water Resources Journal*, *34*(4), 453–466.

Taufik, M., Veldhuizen, A. A., Wösten, J. H.M., & van Lanen, H. A.J. (2019). Exploration of the importance of physical properties of Indonesian peatlands to assess critical groundwater table depths, associated drought and fire hazard. *Geoderma*, *347*, 160–169.

Thompson, D. K., & Waddington, J. M. (2013). Peat properties and water retention in boreal forested peatlands subject to wildfire. *Water Resources Research*, *49*(6), 3651–3658. https://doi.org/10.1002/wrcr.20278

Tiemeyer, B., & Kahle, P. (2014). Nitrogen and dissolved organic carbon (DOC) losses from an artificially drained grassland on organic soils. *Biogeosciences*, *11*(15), 4123–4137. https://doi.org/10.5194/bg-11-4123-2014

Tiemeyer, B., Lennartz, B., & Vegelin, K. (2006). Hydrological modelling of a re-wetted peatland on the basis of a limited dataset for water management. *Journal of Hydrology*, *325*(1-4), 376–389. https://doi.org/10.1016/j.jhydrol.2005.10.039

Timmermann, T., Margoczi, K., Takacs, G., & Vegelin, K. (2006). Restoration of peat-forming vegetation by rewetting species-poor fen grasslands. *APPLIED VEGETATION SCIENCE*, 9(2), 241–250.

Trangmar, B. B., Yost, R. S., & Uehara, G. (1985). Application of geostatistics to spatial studies of soil properties. *Advances in Agronomy. Vol. 38*, 45–94.

Tu, J., & Xia, Z. G. (2008). Examining spatially varying relationships between land use and water quality using geographically weighted regression I: Model design and evaluation. *Science of the Total Environment*, *407*(1), 358–378. https://doi.org/10.1016/j.scitotenv.2008.09.031

Turetsky, M. R., Benscoter, B., Page, S., Rein, G., van der Werf, G. R., & Watts, A. (2015). Global vulnerability of peatlands to fire and carbon loss. *Nature Geoscience*, *8*(1), 11–14. https://doi.org/10.1038/ngeo2325

Urban, J., Ingwers, M., McGuire, M. A., & Teskey, R. O. (2017). *Stomatal conductance increases with rising temperature. Plant Signal Behav 12: e1356534*.

van Bellen, S., & Larivière, V. (2020). The ecosystem of peatland research: a bibliometric analysis. *Mires & Peat*, *26*.

van der Ploeg, M. J., Appels, W. M., Cirkel, D. G., Oosterwoud, M. R., Witte, J. P.M., & van der Zee, S. E.A.T.M. (2012). Microtopography as a driving mechanism for ecohydrological processes in shallow groundwater systems. *Vadose Zone Journal*, *11*(3), 52–62. https://doi.org/10.2136/vzj2011.0098

van der Schaaf, S. (1999). *Analysis of the hydrology of raised bogs in the Irish Midlands : a case study of Raheenmore Bog and Clara Bog* [Dissertation]. RIS. https://edepot.wur.nl/195247

van Dijk, G., Nijp, J. J., Metselaar, K., Lamers, L. P.M., & Smolders, A. J.P. (2017). Salinity-induced increase of the hydraulic conductivity in the hyporheic zone of coastal wetlands. *Hydrological Processes*, *31*(4), 880–890. https://doi.org/10.1002/hyp.11068

Van Gaalen, J. F., Kruse, S., Lafrenz, W. B., & Burroughs, S. M. (2013). Predicting Water Table Response to Rainfall Events, Central Florida. *GroundWater*, *51*(3), 350–362. https://doi.org/10.1111/j.1745-6584.2012.00970.x

Waddington, J. M., Morris, P. J., Kettridge, N., Granath, G., Thompson, D. K., & Moore, P. A. (2015). Hydrological feedbacks in northern peatlands. *Ecohydrology*, *8*(1), 113–127.

Wang, M., Liu, H., Zak, D., & Lennartz, B. (2020). Effect of anisotropy on solute transport in degraded fen peat soils. *Hydrological Processes*, *34*(9), 2128–2138.

Warrick, A. W. (1980). Spatial variability of soil physical properties in the field. In D. Hillel (Ed.), *Applications of Soil Physics* (pp. 319–344). Academic Press.

Webster, K. L., & McLaughlin, J. W. (2010). Importance of the water table in controlling dissolved carbon along a fen nutrient gradient. *Soil Science Society of America Journal*, *74*(6), 2254–2266.

White, W. N. (1932). *A method of estimating ground-water supplies based on discharge by plants and evaporation from soil: Results of investigations in Escalante Valley, Utah* (Water Supply Paper 659A). Washington, D.C. http://pubs.er.usgs.gov/publication/wsp659A https://doi.org/10.3133/wsp659A

Whitfield, P. H., St-Hilaire, A., & van der Kamp, G. (2009). Improving hydrological predictions in peatlands. *Canadian Water Resources Journal*, *34*(4), 467–478.

Whittington, P. N., & Price, J. S. (2006). The effects of water table draw-down (as a surrogate for climate change) on the hydrology of a fen peatland, Canada. *Hydrological Processes*, *20*(17), 3589–3600.

Whittle, A., & Gallego-Sala, A. V. (2016). Vulnerability of the peatland carbon sink to sea-level rise. *Scientific Reports*, *6*. https://doi.org/10.1038/srep28758

Will, R. E., Wilson, S. M., Zou, C. B., & Hennessey, T. C. (2013). Increased vapor pressure deficit due to higher temperature leads to greater transpiration and faster mortality during drought for tree seedlings common to the forest–grassland ecotone. *New Phytologist*, *200*(2), 366–374.

Wilson, D., Blain, D., Couwenberg, J., Evans, C. D., Murdiyarso, D., Page, S. E., Renou-Wilson, F., Rieley, J. O., Sirin, A., Strack, M., & others (2016). Greenhouse gas emission factors associated with rewetting of organic soils. *Mires and Peat*, *17*.

Wittenberg, H., Aksoy, H., & Miegel, K. (2019). Fast response of groundwater to heavy rainfall. *Journal of Hydrology*, *571*, 837–842. https://doi.org/10.1016/j.jhydrol.2019.02.037

Wu, J., & Roulet, N. T. (2014). Climate change reduces the capacity of northern peatlands to absorb the atmospheric carbon dioxide: The different responses of bogs and fens. *Global Biogeochemical Cycles*, *28*(10), 1005–1024.

Yavitt, J. B., Williams, C. J., & Wieder, R. K. (1997). Production of methane and carbon dioxide in peatland ecosystems across North America: effects of temperature, aeration, and organic chemistry of peat. *Geomicrobiology Journal*, *14*(4), 299–316.

Yu, Z., Loisel, J., Brosseau, D. P., Beilman, D. W., & Hunt, S. J. (2010). Global peatland dynamics since the Last Glacial Maximum. *Geophysical Research Letters*, *37*. https://doi.org/10.1029/2010GL043584

Z. Li, & Z. Chen (2012). Comparing two measuring methods of soil microtopography. In *2012 First International Conference on Agro- Geoinformatics (Agro-Geoinformatics)*.

Zare, M., Afyuni, M., & Abbaspour, K. C. (2010). Effects of biosolids application on temporal variations in soil physical and unsaturated hydraulic properties. *Journal of Residuals Science and Technology*, *7*(4), 227–235.

Zauft, M., Fell, H., Glaßer, F., Rosskopf, N., & Zeitz, J. (2010). Carbon storage in the peatlands of Mecklenburg-Western Pomerania, north-east Germany. *Mires & Peat*, 6.

Zeitz, J., & Velty, S. (2002). Soil properties of drained and rewetted fen soils. *Journal of Plant Nutrition and Soil Science*, *165*(5), 618–626. https://doi.org/10.1002/1522-2624(200210)165:5<618::AID-JPLN618>3.0.CO;2-W

Zeleke, T. B., & Bing, C. S. (2005). Scaling relationships between saturated hydraulic conductivity and soil physical properties. *Soil Science Society of America Journal*, *69*(6), 1691–1702. https://doi.org/10.2136/sssaj2005.0072

Zhang, B. J., Zhang, G. H., Yang, H. Y., Wang, H., & Li, N. N. (2018). Soil erodibility affected by vegetation restoration on steep gully slopes on the Loess Plateau of China. *Soil Research*, *56*(7), 712–723. https://doi.org/10.1071/SR18129

Zhang, M., Singh, H. V., Migliaccio, K. W., & Kisekka, I. (2017). Evaluating water table response to rainfall events in a shallow aquifer and canal system. *Hydrological Processes*, *31*(22), 3907–3919. https://doi.org/10.1002/hyp.11306

Zhang, Y.-K., & Schilling, K. E. (2006). Effects of land cover on water table, soil moisture, evapotranspiration, and groundwater recharge: a field observation and analysis. *Journal of Hydrology*, *319*(1-4), 328–338.

Zheng, L., Xu, J., Tan, Z., Xu, L., & Wang, X. (2019). Spatial Distribution of Soil Organic Matter Related to Microtopography and NDVI Changes in Poyang Lake, China. *Wetlands*, *39*(789), 1–13. https://doi.org/10.1007/s13157-019-01131-4

Zhong, Y., Jiang, M., & Middleton, B. A. (2020). Effects of water level alteration on carbon cycling in peatlands. *ECOSYSTEM HEALTH and SUSTAINABILITY*, *6*(1). https://doi.org/10.1080/20964129.2020.1806113

ANNEX: CURRICULUM VITAE OF SATE AHMAD

1. PERSONAL INFORMATION & BACKGROUND

Full Name: Sate Ahmad
Marital Status: single, no children
Age: 33 (DoB: 28 July 1987)
Profession: Research
Nationality: Bangladeshi
Current Address: Petersilienstr. 5, 18055, Rostock, Germany
Email: sate.ahmad@uni-rostock.de; sate.bd@gmail.com
Phone: +4915906802123
RG Profile: https://www.researchgate.net/profile/Sate_Ahmad
WoS Researcher ID:G-8616-2016 | **ORCID**: https://orcid.org/0000-0002-1268-3457
***h*-index: 4**

Research Focus

Wetland ecohydrology

Hydrological restoration

Hydropedology

Fen vegetation

Salinization in Deltas

Background

My interest lies in quantifying ecosystem processes in wetlands and finding out how they are affected by human activities. My current research aims at understanding soil properties and hydrological processes in coastal and non-coastal temperate fen peatland complexes. Additionally, I volunteer for interdisciplinary research on the interactions between human society and the "natural" environment in tropical deltas which include understanding how climate change impacts human well-being through multiple pathways and how ecosystem services are connected to health and livelihoods.

Languages:
Bengali (Proficient); English (Proficient); French (Beginner); German (Beginner)

2. RESEARCH AND TEACHING EXPERIENCE

Jun 2019 onwards
External Collaborator (Volunteer)
Nature's contribution to poverty alleviation, human wellbeing and the SDGs (Nature4SDGs) Project (https://www.nature4sdgs.org/)

Oct 2017 onwards
Research Associate (Wissenschaftlicher Mitarbeiter)
Soil Physics, Faculty of Agricultural and Environmental Sciences, University of Rostock, Germany.
Key Tasks: Research on soil hydrological processes in fen peatlands, under the EU-funded project - WETSCAPES (www.wetscapes.uni-rostock.de/en/).

Jan, 2017 – Aug, 2017
Lecturer (Part-time)
Department of Environmental Science and Management, North South University, Bangladesh

Nov 2016 – Jul 2017
Senior Research Officer
Initiative for Climate Change and Health, icddr,b, Bangladesh
Key tasks: Data analysis, developing research protocol, manuscript preparation

Feb 2016 – Aug 2016
Research Intern (*Stage*)
Department of Applied Ecology and Paleoecology, Christian Albrechts University of Kiel, Germany
Key tasks: Hydrological and vegetation data analysis and thesis report writing

Nov, 2013 – Oct 2016
Research Officer (on study leave from Sep, 2014 – Sep, 2016)
Initiative for Climate Change and Health, icddr,b, Bangladesh
Key tasks: defending research protocol at ethics and research review committee meetings, water sampling (salinity), development of sampling strategy, statistical analysis

Jan 2013 – Oct 2013
Senior Research Assistant
Centre for Population, Urbanisation & Climate Change, icddr,b
Key tasks: developing research protocol and proposal (those related to climate change and environment), questionnaire development, developing sampling strategy

Dec 2011 – Dec 2012
Communication (& Research) Assistant
Centre for Sustainable Development, University of Liberal Arts, Bangladesh (ULAB).

	Key tasks: assisting the coordination of research projects, co-editing a research book published by ULAB & ActionAid Bangladesh
May 2011 – Aug 2011 **May 2010 – Dec 2010**	**Undergraduate Teaching Assistant (Part-time)** Department of Environmental Science & Management, North South University, Bangladesh

3. EDUCATION

Oct 2017 onwards	**PhD Student** Soil Physics, Faculty of Agricultural and Environmental Sciences, University of Rostock, Germany **Topic**: Peatland soil hydrology, WETSCAPES project (www.wetscapes.uni-rostock.de/) **Progress**: 2 articles published, 1 in preparation (see publication section) **Submission:** December 2020, scheduled to be completed by March, 2021. **Associated PhD Student:** DFG Research Training Group (RTG)- Baltic TRANSCOAST (www.baltic-transcoast.uni-rostock.de/)
Sep 2014 – Sep 2016	**MSc. in Applied Ecology (CGPA=4.52/5)** Universities of Poitiers (France), Coimbra (Portugal) & Christian Albrects University of Kiel (Germany) – 100% supported by Erasmus Mundus Master Scholarship (EU). **Thesis:** "Evapotranspiration determination in a restored fen: influence of vegetation composition and management regimes." (Graded "A+")
Jun 2012 – Jul 2014	**Master in (Natural) Resource and Environmental Management (CGPA=3.98/4)** Department of Environmental Science and Management, North South University, Bangladesh – 40% scholarship on tuition fees **Honor**: Summa cum laude
Jun 2007 – Dec 2011	**Bachelors of Science in Environmental Science (CGPA=3.78/4)** Department of Environmental Science and Management, North South University, Bangladesh – 25% scholarship on tuition fees, **Honor**: Magna cum laude

4. RESEARCH AND TECHNICAL SKILLS

- Good knowledge of **R**, **ArcMap**, **QGIS**, **GeoDa**, **Biomapper**, **PAST**
- Proven knowledge of **temporal** & **spatial statistics** (kriging, variograms, etc.)
- Experience in handling **large datasets** (long-term)
- Proficient in **scientific writing** and composing **scientific illustrations**.
- Analysis of vegetation **species diversity** using different indices
- Ecological Niche Factor Analysis (**ENFA**), **NMDS** plotting, **PCA**, etc.
- **Actual evapotranspiration (ET) determination**
- Evaluation of **hydrological functioning** (eg: buffer function)
- Quantifying meteorological controls over **water-table dynamics**
- **Soil** (eg: SOM, K_s etc.) and **water sampling** (eg: pH, EC, TDS, etc.)

5. SELECTED PUBLICATIONS

Peer-reviewed Journal Articles

Ahmad, S., Liu, H., Günther, A., Couwenberg, J., & Lennartz, B. (2020). Long-term rewetting of degraded peatlands restores hydrological buffer function. *Science of The Total Environment*, 141571.

Ahmad, S., Hörmann, G., Zantout, N., & Schrautzer, J. (2020). Quantifying actual evapotranspiration in fen ecosystems: Implications of management and vegetation structure. *Ecohydrology & Hydrobiology*.

Ahmad, S., Liu, H., Beyer, F., Kløve, B., & Lennartz, B. (2020). Spatial heterogeneity of soil properties in relation to microtopography in a non-tidal rewetted coastal mire. *Mires & Peat*, *26*.

Jurasinski, G.; **Ahmad, S.**; Anadon-Rosell, A.; Berendt, J.; Beyer, F.; Bill, R.; Blume-Werry, G.; Couwenberg, J.; Günther, A.; Joosten, H.; Koebsch, F.; Köhn, D.; Koldrack, N.; Kreyling, J.; Leinweber, P.; Lennartz, B.; Liu, H.; Michaelis, D.; Mrotzek, A.; Negassa, W.; Schenk, S.; Schmacka, F.; Schwieger, S.; Smiljanić, M.; Tanneberger, F.; Teuber, L.; Urich, T.; Wang, H.; Weil, M.; Wilmking, M.; Zak, D.; Wrage-Mönnig, N. (2020). From Understanding to Sustainable Use of Peatlands: The WETSCAPES Approach. *Soil Systems*, *4*(1), 14.

Ahmad, S., Liu, Alam, S., H., Günther, Jurasinski, G.., & Lennartz, B. (Under Review). Meteorological controls on water table dynamics in fen peatlands depend on management regime. *Frontiers in Earth Science*.

Signature  **Rostock, 27/08/2020**