Dissertationes Forestales 345

Process-based ecosystem models to support management of drained peatland forests

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Academic dissertation

To be presented, with the permission of the Faculty of Science, Forestry and Technology of the University of Eastern Finland, for public criticism in the auditorium C2, Carelia building, University of Eastern Finland, Joensuu, on the 8th of December, 2023, at 12 o'clock noon.

Title of dissertation: Process-based ecosystem models to support management of drained peatland forests
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Dissertationes Forestales 345
https://doi.org/10.14214/df.345

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ISSN 1795-7389 (online)
ISBN 978-951-651-780-6 (pdf)

Publishers:
Finnish Society of Forest Science
Faculty of Agriculture and Forestry of the University of Helsinki
School of Forest Sciences of the University of Eastern Finland

Editorial Office:
Finnish Society of Forest Science
Viikinkaari 6, 00790 Helsinki, Finland
https://www.dissertationesforestales.fi

Urzainki I (2023) Process-based ecosystem models to support management of drained peatland forests. Dissertationes Forestales 345. 39 p. https://doi.org/10.14214/df.345

ABSTRACT

Forested peatlands are globally significant carbon pools, important forest resources and areas for other bioproduction. Management of drained peatland forests has partly contradictory targets, such as economic profit, climate change mitigation and adaptation, and water protection. Balancing between these targets by comparing different management options requires a thorough understanding of the ecosystem processes, as well as modelling tools that are able to represent their complexity. This thesis presents three such tools, which are based on process-based models, and their applications to concrete water table management scenarios in drained peatlands.

All three works are built upon a common peat hydrological model. Two of the studies analyze the effect of canal-blocking restoration practices on the water table and carbon dioxide emissions in tropical peatlands. The first work shows that using optimization algorithms to choose the location of a fixed number of canal blocks can lead to sizeable improvements on the amount of peat they rewet. The second work systematically analyzes the impact of canal blocks on tropical peatland water tables, and provides insights about their performance for different weather conditions and peat types. The third study presents a peatland ecosystem model focused on the effect of drainage on nutrient dynamics and forest growth in boreal peatlands. By connecting the relevant hydrological and biogeochemical processes, this ecosystem model enables the study of interconnected phenomena such as the identification of the stand growth limiting factor, and the impact of typical ditch network management operations on the nutrient balance and forest productivity.

Keywords: Process-based modeling, peatland hydrology, peatland management, canal-blocking restoration, ditch network maintenance

ACKNOWLEDGEMENTS

Should I base my judgment on my own interaction with other's PhD theses, this could be the most read section of the current work. If this was the case it would be entirely justified, for the individuals mentioned here have made significant contributions to this thesis, each in their unique ways.

Thanks to the University of Eastern Finland (UEF) and to Luonnonvarakeskus (LUKE) for giving me this opportunity.

Thanks to my supervisors, Marjo Palviainen, Hannu Hökkä, and especially to Annamari Laurén, our Finnish mum, for the constant, relentless support and optimism both in science and in life.

Thanks to the unsung heroes: the reviewers of my manuscripts, the scientists on whose shoulders I stand, the creators of all the open-source software I abuse, and all who generously share solutions to difficult computational and scientific problems on the web.

Thanks to all Lukeans with whom I shared a non-trivial chunk of spatiotemporal coordinates around the coffee-machine rooms. And special thanks to whoever included a delay in the hard-drive formatting protocol for old LUKE computers.

Thanks to the wonderful people we met east and south: Marta, Jojo, Marina, Sergio, Aitor, Sakka-Pera, William Edward & Tiina, Siv & Kaj. Those long, unproductive hours spent together were absolutely necessary for finishing the work. And it's your fault that we'll (only sometimes) miss the dark and cold Finland.

Thanks to the Mediterranean-bound friends and family for the love, care, and patience with my thesis-induced existential anxiety and moaning. I don't want to risk missing any names; you know who you are.

Thanks, in particular, to Aita and Ama. Now that I face the wonderful abyss of parenthood myself, I am beginning to appreciate everything you have done—and continue to do!—for me.

And thanks to meru, for everything. You did not write a word, and yet you know your name should appear in the first page, next to mine.

For Lur.
You are not out there yet, but let's make a deal.
I'll teach you all the science I know,
You teach me life.

LIST OF ORIGINAL ARTICLES

This thesis is based on the list of articles below, which are referred to in the text by the Roman numerals I—III. All articles are reprints of previously published articles with the kind permission of the publishers.

I Urzainki, I., Laurén, A., Palviainen, M., Haahti, K., Budiman, A., Basuki, I., Netzer, M., Hökkä, H. (2020). Canal blocking optimization in restoration of drained peatlands. *Biogeosciences*, 17(19), 4769–4784. https://doi.org/10.5194/bg-17-4769-2020

II Urzainki, I., Palviainen, M., Hökkä, H., Persch, S., Chatellier, J., Wang, O., Mahardhitama, P., Yudhista, R., Laurén, A. (2023). A process-based model for quantifying the effects of canal blocking on water table and CO_2 emissions in tropical peatlands. *Biogeosciences*, 20(11), 2099–2116. https://doi.org/10.5194/bg-20-2099-2023

III Laurén, A., Palviainen, M., Launiainen, S., Leppä, K., Stenberg, L., Urzainki, I., Nieminen, M., Laiho, R., Hökkä, H. (2021). Drainage and Stand Growth Response in Peatland Forests—Description, Testing, and Application of Mechanistic Peatland Simulator SUSI. *Forests*, 12(3), 293. https://doi.org/10.3390/f12030293

IU was the main author of **Articles I and II**. In these articles IU developed the computational model and performed the simulations. IU and AL designed the study and analyzed the results. IU wrote the manuscript of **Article I**, and AL, MP, HH and KH reviewed and edited it. KH helped conceptualize the optimization algorithms. AB, IB and MN were responsible for the data collection and provided constant feedback between the model and the data. In **Article II**, IU wrote the manuscript, and AL, MP and HH reviewed and edited it. SP, JC, OW, PM and RY were responsible for the data collection and provided constant feedback between the model and the data. IU helped develop the methodology of **Article III**, together with AL, MP and RL. AL, SL and HH conceptualized the study. AL, SL and KL produced the software. AL, MP and LS validated the model and carried out the formal analysis. The data curation was done by HH. AL and MP wrote the manuscript. All authors contributed to the manuscript review and editing.

Contents

SYMBOLS AND ABBREVIATIONS

Symbols

ζ	Water table, measured from the peat surface [m] (Fig. 3, Eq.(2))
h	Water table from a common reference datum [m] (Fig. 3, Eq.(2))
p	Peat surface elevation from a common reference datum [m] (Fig. 3, Eq.(2))
m_{CO_2}	CO_2 mass flux [$\mathrm{Mg\,ha^{-1}\,yr^{-1}}$]
T	Peat hydraulic transmissivity [$\mathrm{m^2\,d^{-1}}$] (Eq.(8))
S_y	Specific yield, dimensionless (Eq.(10))
P	Precipitation [$\mathrm{m\,d^{-1}}$]
ET	Evapotranspiration [$\mathrm{m\,d^{-1}}$]
h_d	Vertical distance to the block head from a common reference datum (Fig. 3)
y	CWL measured from the canal bed [m] (Fig. 3)
Q	Water discharge through canal [$\mathrm{m^3\,s^{-1}}$] (Fig. 3)
q	Volumetric lateral flow between the peat and the canal per unit length [$\mathrm{m^2\,s^{-1}}$]
n	Manning friction coefficient [$\mathrm{m^{-1/3}\,s}$]
A	Cross-sectional flow area of the canal [$\mathrm{m^2}$] (Fig. 3, Eq.(13))
R	Hydraulic radius [m] (Eq.(20))

Abbreviations

PDE	Partial differential equation
CWL	Canal water level
DNM	Ditch network maintenance
WT	Water table

1 INTRODUCTION

Peat is a type of soil consisting of partially decomposed organic matter which has been accumulated under waterlogged, oxygen-deficient conditions. Peatlands are pieces of the landscape that have accumulated peat over long periods of time.

At a fundamental level, peat is formed when the net mass balance of organic matter is positive, i.e., when the rate of organic matter addition exceeds its loss. Most of the organic matter in peat has its origin in plants, and thus the rate of peat addition is limited by the litter produced by the existing vegetation. Peat loss, on the other hand, occurs primarily via the decomposition of the organic matter. When a peatland ecosystem produces more vegetation litter than it decomposes, peat accumulates. When the decomposition happens faster than the addition of litter, there is a net loss of peat.

Peat water content is a crucial variable controlling the functioning of peatland ecosystems. The amount of groundwater present in the peatland directly affects the mechanisms behind the organic matter creation and loss. In a waterlogged peatland the supply of oxygen is too low for the metabolic needs of plant roots and of the peat aerobic decomposer microorganisms. As a result, both peat decomposition and growth of vegetation slow down. In contrast, drier conditions enhance peat decomposition and vegetation growth.

Several societies around the world have been and continue to be sustained by the economic exploitation of peatlands. Many economic activities, such as forestry and agricultural production, involve removing excess water from the peatland in order to enhance site productivity. The typical procedure consists in either digging new drainage systems or cleaning existing ones in order to provide faster pathways for the water to exit the system.

However, peatland drainage may have negative consequences for the environment. Greenhouse gases—most notably carbon dioxide (CO_2), but also methane (CH_4)—are released either as a side product of the organic matter decomposition or via forest fires, whose likelihood increases after drainage. The carbon (C) that was accumulated over centuries in one of the largest terrestrial carbon pools becomes thus susceptible to being emitted when peatlands become dry. Thus, there is a fundamental tension between the drainage-based economic exploitation of peatlands and its associated environmental impact. Any sustainable economic exploitation of natural resources will need to address this tension and include a strategy for responsible peatland management.

Management of peatland hydrology is the unifying thread of this thesis. The presented articles are all concerned with managing the peat water content through operations in the drainage systems. The common aim of Articles I and II is to evaluate the efficiency of drained tropical peatland canal-blocking[1] restoration practices by means of process-based hydrology models. Article III integrates biogeochemistry and tree growth with the hydrological model to create a boreal peatland forest ecosystem model. With it, cost-benefit analysis of ditch[2] network maintenance (DNM) practices are studied.

[1]The terms *dams* and *blocks* are used interchangeably in the text.

[2]The term *ditch* is often used in the boreal peatland literature, whereas *canal* or *channel* is usually employed for tropical peatlands. Ditches in managed forests tend to be man-made and narrow, while canals might also be natural and are typically wider. Nevertheless, they are indistinguishable regarding their physical function: they are the open water bodies that delineate peat areas. The terms are thus used interchangeably in this text, but we prefer *canal* as the generic term. The only exceptions to this will be the widely-used phrase ”open channel flow”.

1.1 The ecologic and economic relevance of peatlands

Peatlands contain 600–650 Pg carbon (C) in only 3 % of Earth's land surface (Xu et al. 2018), which is twice as much as what the world's tropical rainforest biomass contains (360 Pg), and almost as much as the total C in the atmosphere (750 Pg) (Page and Baird 2016). This fact makes peatlands one of the most important long-term C storages in the planet. And C storage is only one of the many important ecosystem services that peatlands provide, including water regulation, biodiversity protection, and food and fuel resources (Kimmel and Mander 2010).

Peatlands exist throughout a large range of Earth's climatic zones—there are peatlands in tropical, temperate, boreal and subarctic climates. The Global North comprises about two thirds of the world's peatlands by area (Page and Baird 2016). Finland alone contains one third of all the European peat resources: 3.2 Pg C stored over 10^5 km^2 (Päivänen and Hånell 2012). Tropical peatlands, on the other hand, occupy about 10 % or 4.4×10^5 km^2 of the global peatland area, and it could be up to 15 % or 6×10^5 km^2 if peatlands in South America are confirmed to be peat-forming (Xu et al. 2018; Page et al. 2022). Only 16 % (105 Pg) of the total C in peatlands lie in the tropics, but due to their exposure to human and climate impacts, tropical peatlands are nowadays the main source of C emissions to the atmosphere. Tropical peatlands span across 3 continents: Southeast Asia, Africa and Central and South America. Among them, Southeast Asia contains the largest share of C storage—56 % of the tropical peat area and 77 % (68.5 Pg) of the tropical C storage—and it is also where anthropogenic alterations to the peatland ecosystem have been most extensive during the last decades (Miettinen et al. 2016; Xu et al. 2018). Southeast Asian peat is mainly distributed among Indonesia, Malaysia, Papua New Guinea and Brunei, with Indonesia containing the largest share, 65 %. Of the C stored in Indonesian forest biomass, 74 % can be attributed to peatland ecosystems (Page et al. 2011; Page and Baird 2016).

Humans have been using peat for a very long time (Qiu et al. 2021). Our relationship with peat goes at least back to Roman times, when it was used as fuel for house heating (Proulx 2023). A particularly salient example of peatland exploitation in Western history is the case of the Netherlands: starting in the Middle Ages and up to the end of the XIX century, it is estimated that 10 % of the total land surface of the country was lost to the sea due to peat farming (De Decker 2011). Since the 1950s, around 15 million hectares of boreal and temperate peatlands have been drained for forestry and other land uses (Paavilainen and Päivänen 1995). Still nowadays 2000 km^2 of peatlands are used to produce electricity and heating in Finland, Ireland, the Russian Federation, Sweden, Estonia and Belarus (Päivänen and Hånell 2012). But the most widespread use is for forest and agriculture production— it is estimated that 25 % of global peatland area is used for it, and the number possibly doubles if peatlands used as livestock pasture are also included (Page and Baird 2016).

Most of the Finnish drained peatland area, approximately 5.5×10^4 km^2, is devoted to forestry, whereas only 0.3×10^4 km^2 is used for agriculture production (Päivänen and Hånell 2012). A quarter of the national standing volume of forests, or 6.31×10^8 m^3, grows on peatlands (*Forest Resources Statistics [Web Publication]. Natural Resources Institute Finland.* 2020). The amount of drained peatlands has not changed very much in the last decades (Figure 1). The opposite is true for Southeast Asian tropical peatlands, which have experienced a dramatic land use change in the last 30 years (also in Figure 1). In Malaysia, Borneo and Sumatra, during the years 1990–2015, peatland covered by native forest declined from 119,000 km^2 to 46,000 km^2 (79 % and 29 % of the total peatland area, respectively), while managed peatland areas increased from 17,000 km^2 to 78,000 km^2 (11 % and 50 % of the total peatland area) (Miettinen et al. 2016). In other tropical regions, peatland use is currently typically limited to lower-impact activities that do not require drainage. However, there is a considerable threat of future land-use change of those pristine or low-impact-management tropical peatlands that are close to human settlements due to socio-economic pressures (Page and Hooijer 2016).

As it is the case in other endangered ecosystems, the same mechanisms that make the drainage-based bioproduction economically valuable have severe environmental consequences. C loss increases with forest degradation: pristine peatlands are likely to accumulate C, and the emissions from heavily drained forests are larger than the emissions from lightly managed plantations (Page et al. 2022). Not all human disturbances are necessarily harmful for the stability of the peatland ecosystem, yet the kind of economic exploitation that has usually been carried out—including drainage, deforestation and fires—is (Proulx 2023). Peatland drainage, one of the main themes of this work, is economically beneficial because it enables the growth of dryland crops that could not otherwise tolerate the prolonged wet conditions of pristine peatlands. However, multiple studies have reported a long list of negative impacts associated with it. Drainage increases CO_2 emissions (Jauhiainen et al. 2012; Carlson et al. 2015; Ishikura et al. 2018; Novita et al. 2021), the rate of peat subsidence[3], (Hooijer et al. 2012; Evans et al. 2019; Hoyt et al. 2020; Sinclair et al. 2020; Evans et al. 2022), fire risk (Miettinen et al. 2017b; Kiely et al. 2021), nutrient release (Laurén et al. 2021a,b) and export to water courses (Nieminen et al. 2017), and modifies peat physical and chemical characteristics (Könönen et al. 2018).

The magnitude of human impact on peatland ecosystems is best conveyed by looking at its associated CO_2 emissions. Peatlands are responsible for approximately 5 % of all global anthropogenic GHG emissions (Pörtner et al. 2022). The cumulative emissions from peat oxidation alone in Southeast Asian peatlands between the years 1990 and 2015 were 9200 Mt CO_2 (Figure 1(b), Miettinen et al. (2017a)). The total CO_2 emissions including forest fires rises to 700 $\mathrm{Mt\,yr^{-1}}$ (Page and Baird 2016). European peatlands do not stay behind. It is estimated that the GHG emissions from peatlands used for forestry and agriculture in Europe amount to around 360 Mt CO_2 $\mathrm{yr^{-1}}$ (Page et al. 2011).

The good news is that most of these emissions are directly linked to land use. There are thus sizeable opportunities to reduce emissions and to re-establish the self-sustaining nature of peatland ecosystems through changes in human management practices. To know what kind of practices those could be, we first need to understand what are the mechanisms linking peatland drainage to ecosystem services.

1.2 Water table and carbon balance

Despite the wide range of climatic zones in which peatlands exist, the basic mechanisms that create peatland ecosystems are the same. At a fundamental level, peatland ecosystems will accumulate peat whenever the net balance of organic matter is positive, and will lose it when it is negative. Peat, due to its high organic matter content, is composed of around 50 % C, and it is customary to describe the ecosystem level peat accumulation or loss through C balance (Page and Baird 2016). C enters the ecosystem mainly through photosynthetic net primary production, and it is accumulated in living biomass, plant litter and peat. C leaves peatlands primarily in the form of CO_2, both via autotrophic and heterotrophic respiration and as a side-product of combustion in forest fires. In smaller quantities, C flows out of the

[3] The term *subsidence* is usually applied to any lowering of the peat surface, regardless of its cause. The two main causes are peat decomposition, as pointed out in the text, but also peat compaction, more mechanical and more short-term.

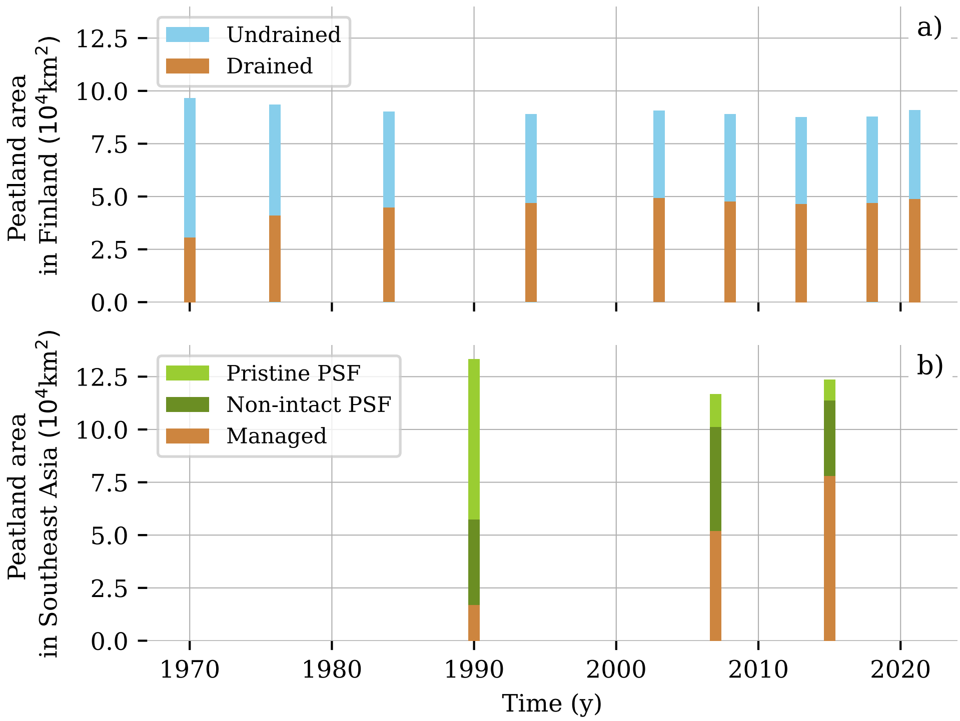

Figure 1: Peatland areas by land use in **a)** Finland and **b)** Southeast Asia (PSF: Peat Swamp Forest). The data for the Finnish case comes from the National Forest Inventory (*Forest Resources Statistics [Web Publication]. Natural Resources Institute Finland.* 2020). The data for the Southeast Asia case (peninsular Malaysia, Sumatra and Borneo) comes from Miettinen and Liew (2010) and Miettinen et al. (2016). The *Non-intact PSF* category includes "all areas where some sort of disturbance has been detected at least once during the three rounds of classification [1990, 2007, 2015]". Thus, following the authors' recommendation, we relabelled the *Degraded PSF* category to the more fitting *Non-intact PSF*. Both industrial plantations and small-holder dominated areas lie under the *Managed* category. Figures **a)** and **b)** may be compared by considering that a *Pristine PSF*, together with a fraction of the *Non-intact PSF* corresponds to the *Undrained* category in the Finnish case, and the remainder to the *Drained* category.

system as CH_4—produced by methanogenic microorganisms—, as dissolved organic carbon and particulate organic carbon (Page et al. 2022). As we will see, a single variable, the water table (WT), greatly influences many of the processes behind peat formation and loss.

1.2.1 Water table and decomposition. First order approximation

The WT is the surface that divides the unsaturated and saturated zones. Virtually all peat pores below the WT are filled with water. Heterotrophic respiration, the main pathway for C loss in peatlands, proceeds at a slow pace when the peat is in waterlogged, anaerobic conditions. Conversely, when the peat is drier and there is enough oxygen supply for decomposer microorganisms, the rate at which C is lost increases. The WT regulates the fraction of peat that is exposed to fast, aerobic decomposition—and, ultimately, whether the peatland ecosystem accumulates or losses peat.

The effect of the WT on long-term C loss has been observed experimentally. There are two common experimental setups to measure peatland C dynamics. One consists in measuring the height of the peat surface over several years. A peat surface that has subsided indicates a net loss of C in the peatland ecosystem. The other experimental method involves directly measuring the soil heterotrophic respiration, one of the main components of the ecosystem C balance, using closed chambers. Experiments employing these two methods have established a relationship between long term WT and C balance. Figure 2 shows the data collected in the review by Carlson et al. (2015) for both subsidence and heterotrophic respiration measurements.

The correlation between WT and C emissions that Figure 2 shows is approximately linear, and this correlation is usually described in the literature using linear models (Jauhiainen et al. 2012; Carlson et al. 2015; Evans et al. 2019; Evans et al. 2022),

$$m_{CO_2}(\zeta) = -a\overline{\zeta} + b. \tag{1}$$

Here m_{CO_2} [$\mathrm{Mg\,ha^{-1}\,yr^{-1}}$] is the mass of emitted CO_2 per unit area and time, ζ [m, negative downwards] is the WT as measured from the peat surface (see Figure 3 for a description of hydrological variables), $\overline{\zeta}$ denotes an annual average of the WT, and a and b are regression coefficients. The negative sign in Eq.(1) implies that the emissions increase with deeper average WT, i.e., more negative $\overline{\zeta}$.

WT has the opposite effect on CH_4 emissions, because methanogenic microorganisms thrive in anoxic conditions. However, the amount of C emitted through methanogenesis is smaller. And although its global warming potential is several times that of CO_2 (Olsson et al. 2019), it will not play a central role in this thesis.

Articles I and II describe the long-term net CO_2 emissions using Eq.(1). Article III, on the other hand, uses an empirical equation for heterotrophic respiration derived for boreal peatland forest stands that, apart from WT, also depends on stand volume, soil temperature and peat bulk density (Ojanen et al. 2010).

1.2.2 Water table, nutrient balance and tree growth. Second order approximation

The effect of the WT on peat accumulation and loss is more complex than our description above might suggest. For one thing, nowhere in our discussion did we include any mention of the primary C input mechanism—organic matter coming from vegetation. When we add plant dynamics to the description of the peatland ecosystem, the WT acquires a new role in it.

The WT can directly suppress vegetation growth—and therefore litterfall and organic

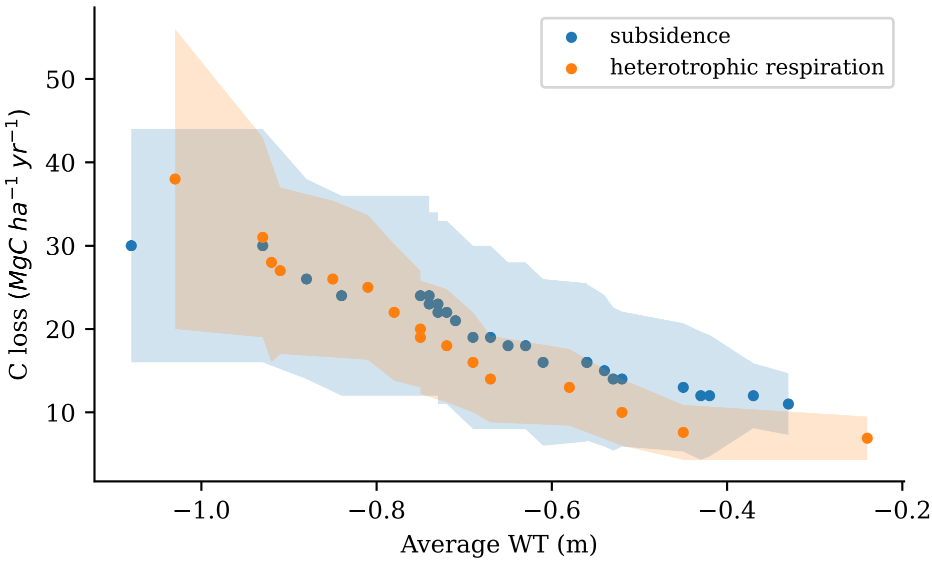

Figure 2: C loss in several Southeast Asian drained peatlands as a function of their annual average WT, as recorded by subsidence measurements (blue) and measurements of CO_2 emissions from heterotrophic respiration using closed chambers (orange). Points show the mean values and the shaded areas indicate the standard deviations. Note that subsidence measurements are a proxy for the net ecosystem C balance, while the heterotrophic respiration chamber measurements only capture one component of this balance. Other components of the C balance such as CH_4 emissions, fluvial export, input from gross primary production, atmospheric deposition, and weathering were not captured by the heterotrophic respiration chamber measurements. The source of the data is Carlson et al. (2015), a review of several studies.

matter input to peat—either by virtue of being too low or too high (Päivänen and Hånell 2012). A very low WT may lead to plant stomatal closure, and consequently to a reduction of plant growth. Conversely, when the WT is too high the root metabolism may be impeded. WT can also indirectly affect vegetation growth through nutrient supply. Organic matter decomposition of peat, apart from generating CO_2, releases nutrients that may be absorbed by the plants. Therefore, the growth of the same forest will be slower with a shallow average WT than with a deep average WT. This indirect effect of WT on peat accumulation is likely to be more pronounced in ombrotrophic peatlands, where nutrient input to the system is limited to rainfall, dusts and marine aerosols, than in minerotrophic peatlands, where nutrients dissolved in external groundwater feed peatlands.

The peatland ecosystem model presented in Article III includes these mechanisms of WT influence on peat development through modelling of the relevant ecosystem processes. Articles I and II, however, do not consider the effect of vegetation on peatland C balance.

1.3 Peatland hydrology

We have established the central role of WT in controlling not only the C dynamics of peatlands, but also the functioning of the ecosystem as a whole. Next, we turn our attention to describing the WT in detail. Ultimately, our goal is to be able to extract insights for better management practices. As we will see, the main human-controllable variable that can influence WT is the canal water level (CWL). This section summarizes the theory that is behind the hydrological models present in Articles I–III.

1.3.1 Groundwater hydrology

The partial differential equation (PDE) that describes groundwater flow in Articles I and II, Eq.(11), is a two-dimensional approximation of the full groundwater flow equation[4]. Equation (11) is sometimes referred to by hydrologists as the Boussinesq equation. In this text we refer to it simply by *the groundwater flow equation*. Understanding the water dynamics that Eq.(11) captures is crucial for our goal of building process-based models of the most important peatland ecosystem dynamics. For a complete derivation the reader is referred to the exhaustive texts Bear (1988) and Bear and Cheng (2010).

First, a word on notation. We have used ζ to denote the WT as measured from the peat surface (see, e.g., Eq.(1)). It is here more convenient to use h [m], the WT height from a reference datum. Both are related by the simple relation

$$h = p + \zeta, \tag{2}$$

where p [m] is the peat surface from a reference datum. These variables are schematically shown in Figure 3.

Let us start by describing the conservation of water mass, a desirable property of any hydrological model. The general conservation equation of a quantity $u(x,y,z,t)$, typically mass or energy, is given by (LeVeque 1992):

$$\frac{\partial u}{\partial t} = -\vec{\nabla} \cdot \vec{f}(u) + s_u, \tag{3}$$

[4]The equation used in Article III, Eq.(24), is a one-dimensional version of the same equation.

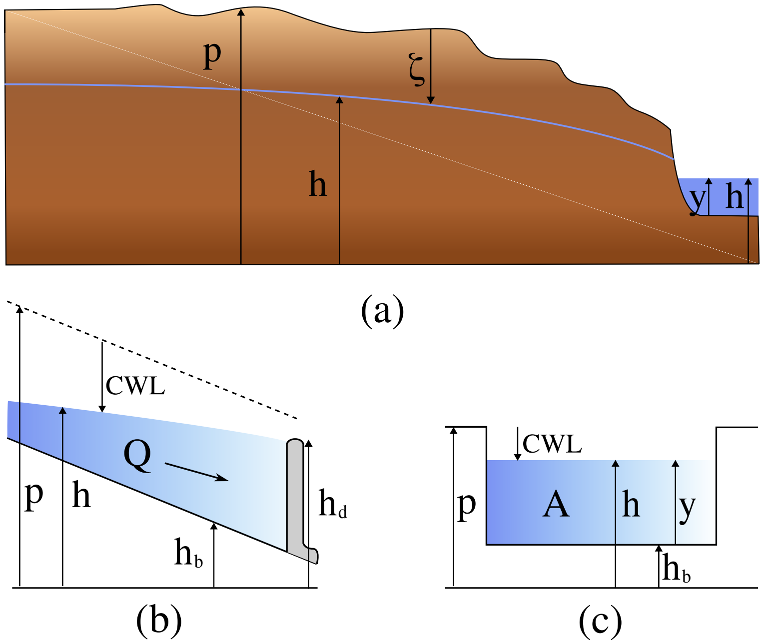

Figure 3: Schematic representation of the hydrological variables. **(a)** A peatland cross-section, bounded in the right by a canal. **(b)** Side view of a canal. **(c)** Canal cross-section.

where $\vec{f}(u)$ is the rate of flow or flux of u, and $s(x,y,z,t)$ is a sink/source term. The amount of u in a given region of space in the time interval $(t, t+dt)$ changes according to the in- or out-flux of u from that region, $\vec{\nabla}\cdot\vec{f}(u)$, and to the amount of u generated or consumed at that region of space, s_u.

Choosing our conserved quantity to be the volumetric water content θ [$\text{m}^3\,\text{m}^{-3}$], Eq.(3) describes the mass conservation of water:

$$\frac{\partial\theta}{\partial t} = -\vec{\nabla}\cdot\vec{q} + s. \tag{4}$$

Here the specific discharge $\vec{q}$ [$\text{m}^3\,\text{m}^{-2}\,\text{s}^{-1}$] describes the volume of water passing through a unit area of porous medium per unit time, and s [s^{-1}] is a possible volumetric water source or sink per unit time.

Henry Darcy, studying the water flow through sand filters in the French city of Dijon, recorded for the first time in the mid XIX century the equation of motion of fluids in a porous medium that now bears his name (Darcy 1856). Darcy's law states that the specific discharge through a given soil volume is proportional to the head difference at the boundaries of the volume, i.e.,

$$\vec{q} = -K\vec{\nabla}H. \tag{5}$$

The proportionality constant K [ms^{-1}] is the hydraulic conductivity, and H [m] is the hydraulic head, containing the suction and gravitational components. In general, the hydraulic conductivity is a tensor. However, in this thesis we assume that the porous medium is isotropic, and the heterogeneity in the peat hydraulic properties will be represented by letting K be a function of the water content of the soil column, $K(\theta)$.

Putting together Eqs. (4) and (5), we now have the complete equation of groundwater flow in three dimensions:

$$\frac{\partial \theta}{\partial t} = \vec{\nabla}\left(K(\theta)\vec{\nabla}H\right) + s. \tag{6}$$

This three-dimensional PDE is, however, seldom used in large scale hydrological models due to the computational resources required. Instead, the Dupuit approximation is usually employed to reduce the description of water flow to a more tractable two-dimensional space. Dupuit was the first to notice that the slope of the WT in many hydrological systems is usually very small (Dupuit 1863). This observation is certainly true of peatlands, where the slopes of the peat surface are very gentle (Anderson 1963; Lampela et al. 2016). The Dupuit approximation therefore consists in assuming that the groundwater flow is essentially horizontal. One may integrate Eq.(6) in the z direction and use the Dupuit assumption to simplify Eq.(6) to a two-dimensional PDE[5],

$$\frac{\partial \Theta}{\partial t} = \vec{\nabla}'\left(T(h)\vec{\nabla}'h\right) + s'. \tag{7}$$

The vertical integration influences all terms of the equation. The gravitational part of $H(x,y,z,t)$ no longer appears explicitly, and it has been replaced by the height of the water table from a reference datum, $h(x,y,t)$ [m]. The conductivity is now replaced by its vertically-integrated counterpart, the transmissivity T [m^{2}s^{-1}],

$$T(h) = \int_{i.b.}^{h} K(z)dz, \tag{8}$$

where *i.b.* stands for the height of the peat column's impermeable bottom.

The source term s' [ms^{-1}] is now measured in units of volume per unit area per unit time. We have also introduced the water storage Θ [m], which is defined as

$$\Theta(h) = \int_{i.b.}^{h} \theta(z)dz. \tag{9}$$

Last but not least, the spatial derivative is now two-dimensional $\vec{\nabla}' = \left(\frac{\partial}{\partial x}, \frac{\partial}{\partial y}\right)$. In what follows, we will drop the prime when referring to the two-dimensional divergence operator: $\vec{\nabla} = \vec{\nabla}'$.

The Dupuit approximation holds whenever the magnitude of vertical flow is much smaller than the horizontal flow, which generally happens in hydrological systems that are much wider than they are thick. In tropical peatlands, the approximation loses accuracy very close to the canal-peat boundary, where the peat surface has steeper slopes and the water movement is more vertical.

The groundwater flow equation in the form of Eq.(7) has an unpleasant property. There are two variables, Θ and h, describing the same quantity, the water content of the peat column. Porous media have the particularity that the fluid can only occupy the fraction of the volume that is not occupied by the solid phase, in this case, soil or peat. This information is incorporated into the specific yield, S_y, which provides a map between Θ and h:

[5]For a more detailed discussion of the errors introduced by the Dupuit approximation, see Connorton (1985). For a derivation of the vertically integrated equations, see Bear (1988) and Bear and Cheng (2010).

$$S_y(h) = \frac{\partial \Theta}{\partial h}. \tag{10}$$

This definition makes the specific yield be the volume of water released from (added to) storage per unit area and per unit decline (rise) of the WT. Equation (7) now may be written in terms of h alone,

$$S_y(h)\frac{\partial h}{\partial t} = \vec{\nabla}\left(T(h)\vec{\nabla}h\right) + P - ET, \tag{11}$$

where $P - ET$ $[\mathrm{m\,s^{-1}}]$, the difference between the precipitation and the evapotranspiration, is the net water input to the system.

This is the PDE used to describe groundwater flow in Articles I and II, and it is also the precursor of the one-dimensional version used in Article III. Equation (11) provides a simplified but precise description of the most important processes affecting how groundwater flows in peatlands. Despite being mathematically two-dimensional the vertical flow is implied in the equation (Connorton 1985), and it has also been called a 2.5 dimensional equation. Solving Eq.(11) gives $h(x,y,t)$, i.e., the WT in the study area through time.

The parameters $S_y(h)$ and $T(h)$ are of particular relevance in Eq.(11). They describe the hydraulic properties of the peat, i.e., how much water the peat can hold and how easily it is conduced. The accuracy of the simulations using Eq.(11) depend, to a large extent, on finding appropriate values of these parameters for the system at hand— without specifying them, Eq.(11) could describe the water flow of any porous material. One of the greatest technical difficulties of solving Eq.(11) lies in the nonlinearity introduced by $S(h)$ and $T(h)$. In particular, $T(h)$ varies in orders of magnitude in the peat column (Kelly et al. 2014; Baird et al. 2017; Cobb et al. 2017; Cobb and Harvey 2019), which forces small timesteps in the computational solution of Eq.(11).

The focus of this thesis is on modeling the impact of management practices at the canals or ditches. Nowhere in this equation do we find, however, any mention of the CWL. That is because canals are not part of the peat, the domain where the Eq.(11) describes water flow. Rather, the water levetl at the open water bodies surrounding the peat are the boundary conditions of this PDE. In the next section, we describe this missing part of the puzzle.

1.3.2 Open channel flow

As was the case in the derivation of the groundwater flow equation, the open channel flow equations—also called Saint Venant equations after their discoverer (Saint-Venant 1871)—consist of the mass conservation equation and the equation of motion, or momentum equation. Our derivation of the open channel flow equations will sacrifice mathematical rigor in favor of comprehensibility. For a complete, detailed derivation of the equations the reader is referred to Cunge et al. (1980), Novák (2010), and Szymkiewicz (2010).

The conservation of mass—divided by the density of water— has the shape of the general conservation equation, Eq.(3),

$$\frac{\partial A}{\partial t} = -\vec{\nabla}\cdot\vec{Q} + q. \tag{12}$$

Here, A $[\mathrm{m^2}]$ is the cross-sectional flow area, $\vec{Q}$ $[\mathrm{m^3\,s^{-1}}]$ is the discharge, and q $[\mathrm{m^2\,s^{-1}}]$ is the lateral flow per unit length, which functions as a sink/source term that accounts for the water flow in the interface between the peatland and the canals.

Water flows primarily along the longitudinal direction of the canal, and Eq.(12) can be safely simplified to a one dimensional description. Such simplification assumes that water movement in the directions perpendicular to the canal (both horizontal and vertical) are negligible[6]. The cross-sectional flow area, A, depends on the canal cross-sectional geometry and the water height from the canal bed, y [m] (see Figure 3). In the absence of better sources of information, our model in Article II worked with the assumption that the canal cross-section was rectangular. This gives the simple relationship

$$A(y) = By, \tag{13}$$

where B [m] is the (possibly varying) width of the rectangular canal cross-section.

In the groundwater flow equations, Eq.(11), we used the water height from a common reference datum, h, as the water stage variable. Equation (13), however, is written in terms of

$$y = h - h_b, \tag{14}$$

where h_b [m] is the height of the canal bed from a common reference datum (Figure 3). The coupling between groundwater and open channel flow models is simplified when the same variables are used. So, for rectangular canals, one may write

$$\frac{\partial A}{\partial t} = B\frac{\partial y}{\partial t} = B\frac{\partial h}{\partial t}, \tag{15}$$

where we have used the fact that h_b does not vary over time.

Thus, the one dimensional mass conservation version of Eq.(12) becomes

$$\frac{\partial h}{\partial t} = -\frac{1}{B}\frac{\partial Q}{\partial x} + \frac{q}{B}, \tag{16}$$

where x is the direction of the canal.

Equation (16) is the first of the two equations that describe open channel flow. The second equation comes from applying Newton's second law: the change of momentum of water is equal to the sum of external forces acting on it. This results in the following expression:

$$\frac{\partial Q}{\partial t} = -\frac{\partial}{\partial x}\left(\frac{Q^2}{A}\right) - gA\frac{\partial h}{\partial x} - gAS_e, \tag{17}$$

where g [m s^{-2}] is the gravitational acceleration, and S_e is the so-called friction slope.

The first two terms describe the change in momentum and its flux. The third term contains two external forces,

$$\frac{\partial h}{\partial x} = \frac{\partial y}{\partial x} + \frac{\partial h_b}{\partial x}, \tag{18}$$

which stand for the pressure and the canal bed reaction forces, respectively. The last term, $-gAS_e$, describes friction forces in the direction of water flow. The friction slope, S_e, is usually assumed to be proportional to the square of flow velocity, and there exist several empirical expressions to model it. In this work we choose the Manning friction slope,

[6]The one-dimensional flow simplification loses accuracy in meanders, where the direction of water is not aligned with the direction of the canal. As was the case with the two-dimensional simplification of the groundwater flow, however, the one-dimensional approximation is considered a good accuracy-efficiency trade-off, especially in large canal networks. For further information on the comparison between one- and higher-dimensional models of the open channel flow equations, the reader is referred to Ghostine et al. (2012).

$$S_e = n^2 \frac{|Q|Q}{A^2 R^{4/3}}, \tag{19}$$

where we have used $|Q|Q$ instead of Q^2 to preserve the sign of the flow direction, and R [m] is the hydraulic radius, defined as the quotient between the cross-sectional area, A, and the wetted perimeter. In the simple case of the rectangular channel, R is given by

$$R = \frac{A}{2y+B}. \tag{20}$$

Several assumptions underlie Eq.(17). Most prominently, we are assuming that the velocity of water is uniform over the cross-section, that the effects of boundary friction and turbulence can be accounted for through empirical friction laws, and that the friction forces act only in the direction of water flow.

Together, Eqs.(16) and (17) form the set of open channel flow equations, which are reproduced here for clarity:

$$\begin{cases} \dfrac{\partial h}{\partial t} = -\dfrac{1}{B}\dfrac{\partial Q}{\partial x} + \dfrac{q}{B} \\ \dfrac{\partial Q}{\partial t} = -\dfrac{\partial}{\partial x}\left(\dfrac{Q^2}{A}\right) - gA\dfrac{\partial h}{\partial x} - gAn^2\dfrac{|Q|Q}{A^2 R^{4/3}}. \end{cases} \tag{21}$$

In Article II we used an approximation that simplifies the description of water dynamics further. This approximation, called the diffusive wave approximation, is obtained by neglecting the first two terms of the momentum equation from Eqs.(21), that is,

$$\begin{cases} \dfrac{\partial h}{\partial t} = -\dfrac{1}{B}\dfrac{\partial Q}{\partial x} + \dfrac{q}{B} \\ \dfrac{\partial h}{\partial x} = -n^2\dfrac{|Q|Q}{A^2 R^{4/3}}. \end{cases} \tag{22}$$

Neglecting these terms amounts to stating that the change in water velocity in time and space are small relative to the magnitude of the pressure, slope and friction forces.

In the presence of a dam, only the first of the two equations in Eq. (21) is valid: The conservation of mass still applies, but the canal block changes the forces affecting the flow of water. A simple way to model the effect of dams is to replace the momentum equation with an expression for the water discharge through them, Q_d [m^3 s^{-1}] (Szymkiewicz 2010),

$$Q_d = K_d\,(h - h_d)^{1.5}. \tag{23}$$

Here, K_d [m$^{3/2}$ s^{-1}] is a coefficient regulating the rate of water flow through the block that needs to be determined experimentally, and h_d [m] is the elevation of the block head above the reference datum (see Figure 3).

1.3.3 Management of peatland hydrology

We now have a complete mathematical description of the water flow in the peatland system: Eq.(11) models the WT, and the system of Eq.(21) or (22) models the CWL. Yet our interest is beyond mere understanding—we also wish to control the WT so as to restore the peatland ecosystem. Let us investigate the possibilities by looking at the equations.

Equation (11) suggests a straightforward way to raise WT: increase the net water input,

$P-ET$. Another, more subtle way, is to modify the peat hydraulic properties T and S_y so that more water is retained in the system. These are precisely the levers that revegetation restoration practices pull (Yuwati et al. 2021). A successful revegetation project may be able to change the peat hydraulic properties and the net external water input budget at scale.

The articles in this thesis are concerned with a more direct approach to control WT: managing the water level at the open water bodies surrounding the peat. Looking at Eqs.(21) and (23), there is an obvious way in which we could manage the CWL. Building tall (large h_d) dams that let small amounts of water through (small K_d) will raise the CWL and, in turn, raise the WT. This is the approach studied in Articles I and II: CWL management through dam-building. That is not the only way to affect the CWL; similar effects might also be obtained either by changing the geometry of the canal cross-section—which would affect $A(y)$ and $R(y)$—or by directly modifying the Manning friction coefficient n. Over time, canals might become clogged with vegetation and other organic materials, and their cross-sectional shape might vary due to sedimentation, vegetation ingrowth, and erosion. Both modify the friction of water flow, controlled by S_e, and, as a consequence, they change the CWL. Ditch network maintenance (DNM) operations are a set of practices that are usually employed in boreal peatlands to maintain the CWL at the target levels by cleaning excess vegetation and modifying the canal width and cross-section shape. Article III studies the shift in CWL due to the DNM operations in Finnish peatlands[7].

1.4 Study aims

This work seeks to advance the understanding of responsible management practices of peatland ecosystems by using process-based models. Articles I and II are concerned with canal blocking restoration practices in tropical peatlands and their hydrological description. Each one tries to answer a different management question. Article I studies the impact of block locations in the success of the restoration by exploring the effectiveness of algorithmically designed canal block layouts. Article II analyzes the impact of canal blocks in an existing restoration project.

Article III, on the other hand, presents a complex peatland ecosystem model built on top of the hydrology model, that can be used to simulate the stand growth under different management scenarios. Its complexity is necessary to understand how different processes, such as the effect of ditch water level on the forest's nutrient cycle, interact. With it, we evaluate the impact of DNM scenarios on peatland forest growth and other ecosystem variables that are relevant for management planning.

2 MATERIALS AND METHODS

Articles I and II are concerned with tropical peatland canal-blocking restoration efforts, while Article III studies boreal peatland forest management. The hydrological model is a common skeleton shared by the models presented in all articles. The aims of Articles I and II required a spatially-explicit hydrological model. Catchment-scale models of peatlands need to encompass areas of different canal structure—from managed areas with regular canal grids to pristine areas with few or no canals. On the contrary, managed boreal peatland forests tend to be

[7] It should be noted, however, that Article III does not model the CWL dynamically—it does not use Eq.(21), and the CWL shift due to the DNM is introduced directly.

delimited by ditches laid out in a fishbone pattern, repeated regularly across the landscape. When modeling these systems, one may abstract one of the spatial dimensions because the relevant dynamics are already captured in one of the spatial dimensions—that is why Article III has a 1D hydrological model.

All three works analyze the impact of management practices at canals. Articles I and II consider canal blocking restoration practices, and Article III studies DNM practices. All three studies were conceptualized as tools to be used by stakeholders to inform their decision-making process, and thus the models are all based on easily available field data and on open source software.

2.1 Study areas

Articles I and II each analyzed a different Indonesian peatland area. The areas are located in Sumatra, separated by about 300 km of each other. The study area of Article I is located in the Riau province and it spans 1100 km^2, 931 km^2 of which are covered by peatlands. Article II studied a 220 km^2 peatland area part of the Sumatra Merang Peatland Project, an ecosystem restoration concession located in South Sumatra. Peat depth was similar in both sites—ranging approximately from 2 m to 8 m and averaging at about 5 m. Given their proximity, the climate and general hydrology is similar in both areas[8].

The canal network of Article I was longer than the one in Article II—1100 km versus 219 km. Both canal networks are topologically heterogeneous and have reaches of varying width and depth. Due to the different goals and study designs of each work (see below for more details), the maximum number of canal blocks considered for the study area of Article I was 80, while the restoration project of Article II contained 168 peat compaction dams. This resulted in an expected block density of one block every 14 km in Article I and one block every 1.3 km in Article II.

Article III studied a total of 218 stands from 74 different Finnish drained peatlands. Most of the sites were located in central Finland, although there were some in norther locations. The sites' tree stands were dominated by Scots pine (*Pinus sylvestris* L.), and spanned a broad spectrum of site fertility classes (2 through 6), initial stand volumes, peat types and stages of decomposition, and initial peat nutrient contents.

2.2 Hydrology modelling

The main properties of the hydrological models of Articles I–III are summarized in Table 1.

2.2.1 Groundwater hydrology

The peatland groundwater hydrology was simulated using the Dupuit approximation of the groundwater flow equation. Unlike the large and heterogeneous tropical peatland areas studied in Articles I and II, the Finnish sites of Article III were delineated by regularly arranged ditch networks. The ditches delimited narrow (between 25 m and 60 m) and long (more than 200 m) rectangular peatland areas. In such sites drainage WT dynamics are dominated by the

[8] The study area in Article I borders the Strait of Malacca, which could have hydrological consequences in the form of tidal effects. However, no such effects were observed and the boundary with the sea was treated identically in hydrological terms.

Table 1: Properties of the hydrology models of Articles I–III.

Groundwater flow			
Article	PDE	study area size	mesh
I	Eq.(11)	1100 km^2	rectangular grid
II	Eq.(11)	220 km^2	triangular mesh
III	Eq.(24)	25–60 m $\times$ 200 m	1D equispaced grid
Open channel flow			
Article	model	Canal network length (km)	Number of blocks
I	static	1100	0–80
II	dynamic, Eq.(22)	219	168
III	static	–	DNM, no blocks

nearest parallel ditches. That is why, in this study, each site was conceptualized as a peat strip delimited by the nearest parallel ditches at the boundaries. The regularity of the study areas allowed Article III to use the one dimensional version of Eq.(11), i.e.,

$$S_y(h)\frac{\partial h}{\partial t} = \frac{\partial}{\partial x}\left(T(h)\frac{\partial h}{\partial x}\right) + s, \tag{24}$$

where s [$\mathrm{m\,d^{-1}}$] is the external water source term.

In Articles I and III the mesh was regular—a two- or one-dimensional rectangular grid, respectively. In Article II an unstructured triangular mesh was used.

The source terms of Eq.(11) and (24) were computed using different approaches in the three articles. In Articles I and II precipitation, P, was adopted from on-site weather measurements. Evapotranspiration, ET, was set to a constant for Article I and part of Article II, where a Penman-Monteith equation was also used. In Article III, the source term was computed using the above ground water budget of the SpaFHy hydrology model (Launiainen et al. 2019).

2.2.2 *Open channel flow*

The CWL was modelled with varying degrees of sophistication in the different studies. The description of canal water level in Articles I and III is static: there is no model of the open channel flow dynamics. In these studies the CWL was informed either by external data or by some concrete modelling choice, and the CWL was treated as the boundary condition for the groundwater flow PDE—a Dirichlet boundary condition in Article I and a mix of no-flux conditions and Dirichlet conditions, depending on the state of the WT with respect to the CWL, in Article III. For Article II, however, a custom dynamic open channel flow model based on the diffusive wave approximation of the open channel flow equations, Eq.(22), was created. The model was developed according to the following criteria. First, it should be able to represent the influence of blocks on CWL. Second, so as to increase its reusability, it should be easily applicable to different canal networks. Third, it should be efficient enough to run in large, interconnected canal networks. The details of the resulting model are given in Appendix A of Article II. This dynamic open channel flow model allows for more realistic catchment scale, long-term hydrological simulations.

In all three articles, CWL management scenarios were analyzed. Articles I and II simulate the effects of canal blocks, and Article III models the DNM. In Article I, the CWL was made to instantaneously raise to the block head level in the presence of a block. The resulting CWL propagated upstream to the fixed level (see Figure 4 in Article I) and was static. Similarly, in Article III the CWL was set manually at different heights (-0.3 m and -0.9 m for the DNM scenario calculations). However, the modeling domain of Article III was a one-dimensional peat strip, and unlike in Article I, there was no notion of upstream parts of the canal network. In Article II, the CWL was dynamic and the blocks were modelled with Eq.(23). With this approach, the CWL at both the upstream and the downstream directions is affected by building a block.

2.3 Study design

This section provides an overview of the experimental design adopted to try to answer the main questions of Articles I–III. For a more detailed description, the reader is referred to the Materials and Methods section of each article.

2.3.1 Article I

The following question motivated and guided this work: which block locations maximize the volume of rewetted peat, given a finite number of blocks? This is a recurrent question when planning canal blocking peatland restoration projects. In order to answer it, we coupled a hydrological model to a series of optimization algorithms. The optimization algorithm proposes locations for the blocks and the hydrological model finds the WT and the amount of rewetted peat resulting from that block configuration.

The hydrological model consisted of two modules. One module computed the CWL resulting from building canal blocks at given locations in the canal network; the other solved the peatland WT corresponding to that CWL. The CWL model was geometrical rather than physical. Instead of describing the water dynamics, it automatically modified the CWL upstream of a block to match the block head level. Using the computed CWL as boundary conditions, the groundwater model solved the PDE of Eq.(11) to find the WT. The groundwater model was run for 3 days without rainfall and a constant evapotranspiration, starting from completely water-saturated conditions. Thereafter, the total volume of peat above the WT was computed.

The reason for choosing a simple CWL model, rather than a dynamic open channel flow model as in Article II, was the trade-off between the accuracy of the hydrological models and the efficiency required for searching the space of possible block configurations.

We analyzed and compared the performance of five different methods for choosing block locations: random block locations, a rule-based approach that tried to mimic the current methods in restoration practices, and three optimization algorithms. Two of the optimization algorithms, based on simulated annealing and the genetic algorithm, treated the WT as the target function. The other used the genetic algorithm to target the maximum rise of the CWL instead. The total number of blocks ranged from 5 to 80. For each of those numbers of blocks, several sets of block locations were tested for each method: 2000 sets of random locations, 1 set of the rule-based approach (only for 5 and 10 blocks), 6000 sets as suggested by the simulated annealing, 60000 sets as suggested by the genetic algorithm targeting the WT, and 2.5 million sets as suggested by the genetic algorithm targeting the CWL.

As a means to put the potential environmental impact of the different canal blocking strategies into perspective, the annual CO_2 emissions corresponding to the best available block configuration was compared to the unblocked case. For this, we simulated the WT for one year and the resulting CO_2 emissions were computed with the linear relationship of Eq.(1).

2.3.2 Article II

The goal of this work was to evaluate the effectiveness of canal blocking practices as a means to raise the WT in large tropical peatland areas. The hydrological model integrated a realistic simulation of the open channel flow with the groundwater module of Article I. The hydrological model was applied to an Indonesian peatland area undergoing a canal blocking restoration project with a total of 168 blocks. Precise weather and WT measurements were available for this area. In order to account for the large variability inherent to tropical peatland systems, we considered four different scenarios: A very dry and a very wet year, reflecting the range of rainfall in the tropics, as well as low and high values for the peat hydraulic conductivity, which experiments suggest may vary in orders of magnitude from site to site (Baird et al. 2012; Cobb et al. 2017). The WT was simulated with and without blocks for each weather and peat hydraulic property scenario, allowing us to extract the raise in WT due exclusively to the canal blocks.

The plausibility of the model was evaluated by comparing the resulting WT with the measurements of 141 dipwells over a year. For this reality check, the hydrological model was run using precise locally measured weather data.

The development and testing of the dynamic open channel flow model was an additional important goal of this work. This model introduced some novelties that simplify the computation of the open channel flow equations. We used the conservation of mass of Eq.(3) together with a diffusive wave approximation to simplify the computation at the channel network junctions.

Finally, using the same linear relationship as in Article I, Eq.(1), we estimated the reduction in CO_2 emissions due to raised WT. Thus, the effectiveness of canal blocking for drained tropical peatland emission reduction could be analyzed.

2.3.3 Article III

The main aim of Article III was to present a mechanistic model designed to guide the search for better boreal peatland forest management practices. In order to meet this goal, the model embedded the growth response of a stand into the hydrologic description of a given peatland area, while incorporating all the relevant feedbacks between the different processes. The WT, together with peat temperature, affects the rate of organic matter decomposition, which was computed via an empirical heterotrophic respiration model (Ojanen et al. 2010). It also restricts the net primary production in drought or water saturation conditions. The nutrient concentration of the decomposed peat determines the new available nutrients (nitrogen, N, phosphorus, P, and potassium, K) that are added to the pool. Canopy and ground vegetation compete for those nutrients. The potential stand volume was computed according to both the net primary production and the nutrient availability using empirical stand nutrient content models. The realized stand volume was set to the minimum of those two potential volumes by following Liebig's law of minimum, allowing the identification of the growth limiting factors.

The dataset used for the model development was separated from the dataset in which the model was tested so as to minimize potential biases. Modelled WT and stand growth were compared in the two datasets. A sensitivity analysis was carried out with the development dataset in order to infer the sensitivity of the model results to parameters and variables with large uncertainties and to parameters relevant in DNM practices. To showcase the usefulness of the model as a tool to inform practical management questions, two different DNM scenarios were studied in the testing dataset: deteriorated ditches, corresponding to a CWL at -0.30 m, and cleand ditches, corresponding to CWL at -0.90 m. The effects of these DNM management scenarios in the WT and in the growth of the testing dataset stands were analyzed.

3 RESULTS

It could be argued that the most important results of Articles I, II and III lay not in any particular output of the computational models, but in the models themselves. For instance, Articles I and II contain several observations about the effectiveness of canal blocks in two particular tropical peatland areas, and yet the most scientifically impactful outcome may very well be the development and testing of the presented models, since they are tools that may be applied to many more areas and situations than we did in our articles. In that sense, the models are generalizable and more useful to real-world peatland management applications than the specific results that we presented might indicate.

Nowhere is this more evident than in Article III. While in Articles I and II relatively simple models were used to tease out as many insights as possible on a given research question, Article III was more focused on the description of its comparatively complex model than on any application. The main result of Article III was thus the development and testing of a process-based boreal peatland ecosystem model that aims to capture the most relevant mechanisms relating stand growth, peat nutrient availability, decomposition, WT and CWL. Thus, it facilitates new ways for quantitatively reasoning about otherwise elusive peatland ecosystem processes.

Having made this caveat, we now present a summary of the findings that were made by applying the models to answer the concrete questions described in the study aims.

Raising the CWL also raised the average peatland WT under all the studied hydrological conditions both in the tropical and boreal peatlands (Articles I, II and III). Specifically, a net WT rise was observed when comparing the situation with and without blocks for different block locations, different total number of blocks, different peat hydraulic properties and different weather conditions in tropical peatlands (Articles I and II). Lower WTs were also observed in Finnish peatlands as a result of the lower CWL corresponding to DNM practices (Article III).

A corollary of the previous result, canal blocks led to a decrease of the net CO_2 emissions (Articles I and II). Building 80 blocks over the whole 931 km^2 peatland area of Article I mitigated 2.2 % of the CO_2 that would have been emitted without blocks. The 168 blocks that were built in the 220 km^2 area of Article II, on the other hand, mitigated 2.5 % of the CO_2 emissions, averaged over all modelled scenarios.

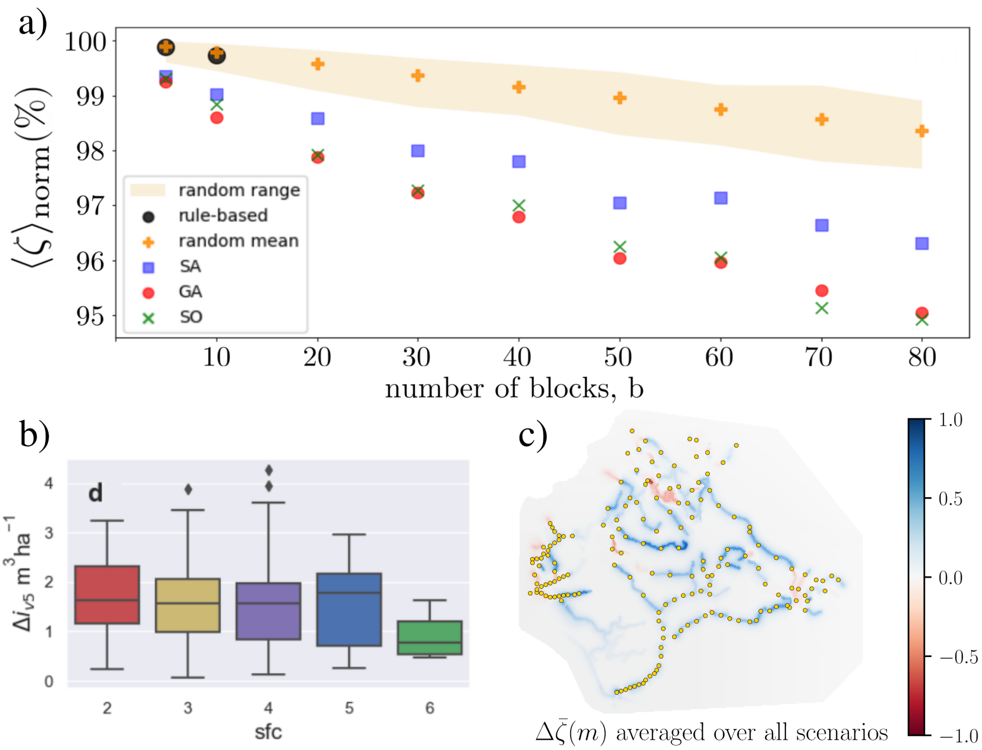

Figure 4: A collection of figures from Articles I–III, showcasing the most important results. **a)** From Article I. Percentage of rewetted peat for different numbers of blocks, for all different block locating methods (SA: simulated annealing, GA: genetic algorithm, SO: simple optimization). **b)** From Article III. Increase in the five-year volume growth as a result of lowering the CWL from -0.3 m to -0.9 m, for each site fertility class. **c)** From Article II. WT difference between the blocked and unblocked cases, averaged over all modelled scenarios. Areas in red correspond to lower WT in the blocked case. Yellow dots show the dam locations.

Nevertheless, the effect that CWL management operations had on the peatland WT was smaller in magnitude than the effect of the annual rainfall variability. No amount of canal blocks or DNM operations were able to keep a shallow average WT during dry periods (Articles I, II, III).

While canal blocks were observed to raise the overall WT, they did not do so everywhere in the study area. In fact, they were found to *lower* the WT in some areas compared to the unblocked case (Article II, Figure 4 c)). Canal blocks did not significantly affect the WT further than 600 m inside the peatland in any of the modelled scenarios (Article II). Therefore, assessing block performance by using dipwells that are close to canals might overestimate their effect on the WT. Additionally, it was observed that a larger peat hydraulic conductivity results in WTs that are more sensitive to changes in the CWL. Larger hydraulic conductivity values increase the distance to which the CWL signal propagates into the peatland, and the average WT changes are greater as a result.

As a general rule, the more blocks were installed in a given area, the larger the total volume of peat they rewetted. This was true even for blocks that were located randomly (Article I, Figure 4 a)). All the optimization algorithms presented in Article I were able to find block locations that rewetted substantially more peat than randomly located blocks. The human rule-based approach for locating blocks, on the contrary, did not outperform the randomly located blocks. The best results of the block location optimization algorithms rewetted between 3 and 7 times more peat volume than the randomly placed blocks. All algorithmic block placement methods saw a decrease in effectiveness as the total number of blocks increased. This was not observed for randomly located blocks. This effect was likely due to the increasing computational complexity of the optimization problem, and not to the number of blocks reaching the limits of the finite-sized area.

In Article III, the DNM ditch-deepening operations, which were modelled by a decrease in CWL from -0.3 m to -0.9 m, had a positive effect in the stand volume growth for all studied sites in Finland (Figure 4 b)). The stand volume growth due to DNM operations was comparable in all but the least fertile stands, where the five-year average stand volume growth was significantly smaller. Low K availability was identified as the main growth-limiting factor in most of the studied Finnish stands. A few sites were limited by physical constraints instead.

4 DISCUSSION

4.1 Impact of canal water management on water table

The three works presented in this thesis consider how management operations in the open water bodies that delimit tropical and boreal peatland areas may affect the WT, and, through it, other ecosystem processes. In tropical peatlands that are targeted for restoration the typical management operation consists in blocking the canals to raise the overall WT throughout the peatland area. In boreal production-oriented peatlands, on the other hand, ditches are cleaned via DNM to lower the WT and thus enhance stand productivity. They are two sides of the same coin: in the end, canal blocks and DNM operations result in a higher or lower CWL which, in turn, modifies the WT.

A peatland catchment may be considered, from a bird's eye view, as a system with a certain total inflow and outflow of water. Building canal blocks or leaving the ditches uncleaned both act as an abstract friction term between the incoming and outgoing water. Blocks increase the residence time of water inside the peatland system and, therefore, must raise the WT compared to the unblocked or the clean-ditch scenario. Mathematically, this follows from a straightforward observation about the groundwater flow equation, Eq.(11). CWL acts as the boundary conditions of that PDE, and higher values of the boundary condition enforce higher values of the solution variable (WT) inside the equation domain.

And yet, at the same time, raising CWL in some areas may not result in a higher CWL throughout the canal network. This follows from the mass-conservation of water and the connectivity of the canal network. Blocking water at one point implies that the downstream parts of the canal will receive less water. Areas of the peatland with a certain combination of canal network topology and peat surface topography may end up with lower WT as a result. Similarly, although it has not been observed in any of the articles, the converse should also be true: lowering the CWL in some areas—either by removing blocks or cleaning the ditches—may result in higher CWL downstream in the canal network.

The magnitude of the observed net positive effect that CWL management practices have on raising WT is smaller than the effect that weather has on WT. This is obviously true when the peatland is dry enough that the canals are empty: once canals are dry, canal blocks or uncleaned ditches have no impact on WT (Putra et al. 2021, 2022). But it is also true in general. Figure 10 in Article I and Figure 7 (a,b) in Article II both show how little influence raised CWLs have on the overall WT. Specifically, Figure 7 in Article II provides a side by side comparison of the blocked and unblocked average WT in two extreme weather conditions in the tropics. It is apparent that, at least in very dry conditions, canals blocks are powerless in raising the overall WT. This is especially true when the peat hydraulic conductivity is low. The difference that the peat hydraulic conductivity makes in raising the WT can also be seen in Figure 7 in Article II. Hydraulic conductivity measures the ability of the peat to conduct water, and thus it is connected to the speed and the length at which the signal from the canal propagates into the peat.

The magnitude of the effect of CWL on WT also depends on the canal network density. A larger density of canals such as the Finnish peatlands studied in Article III allows for a greater control of the WT through operations in canals. In these stands, the parallel ditches were between 25 m and 60 m away from each other. The sensitivity analysis carried out in Article III showed that WT was quite sensitive to how far apart the ditches were under these conditions (see S_{width} values in Figure 5, Article III).

It is important to mention, at the same time, that CWL management practices raise WT in the areas where subsidence happens faster, i.e., near canals (Evans et al. 2022). In Article II we found that canal blocks had an overall positive effect in raising the WT during the first year after installation within 600 m from the canals. Further away, the blocks had no impact on the first year WT, even when the peat had an hydraulic conductivity in the top range of the values found in the literature (Kelly et al. 2014; Baird et al. 2017; Cobb and Harvey 2019).

The limited distance to which operations in the canals affect the WT suggests that one should be cautious when interpreting empirical measurements of WT, particularly in tropical peatlands. Tropical peatland canals, apart from having the effect of lowering the WT by removing water from the system, also function as pathways for travel and for wood transportation through the otherwise inaccessible peatland forest (Dohong et al. 2017). That is why the WT measuring dipwells are usually located near canals in tropical peatlands. Given the result above, one should be careful when inferring the restorative effects of canal management practices by only looking at the dipwell measurements: there is a risk of overestimation when extrapolating the measurements far away from canals.

The different canal blocking densities of Articles I and II and the corresponding volume of rewetted peat also deserve some commentary. According to Table 1, the maximum canal block density in Article I was one block every 13.8 km (or 80 blocks spread throughout 1100 km of canals), while in Article II it was one block every 1.3 km (or 168 blocks in 219 km of canals). However, our results show that the percentage of peat volume that those blocks rewetted as compared to the unblocked case was 2.2 % in Article I, and 2.5 % in Article II[9]. How can a ten times more dense canal block layout result in practically the same amount of rewetted peat? The answer lies in a combination of two factors. The first is the accuracy of the different open channel flow models. In Article I, the CWL was static and constant throughout the modelled year, and so it was unaffected by the vagaries of climate. Moreover, the response of the CWL to placing a block was such that it immediately changed the CWL upstream, but did not do so downstream. Therefore, not only does the model of Article I fail to capture water dynamics, but neither does it respect conservation of mass: water in the upstream just materializes out of thin air[10]. The CWL model from Article II, on the contrary, is more physical, and it is the one that should be considered to derive conclusions and or to inform management practices. The second factor is that in Article I the blocks were chosen to maximize modelled WT, while the ones in Article II come from an existing restoration plan. The block locations of Article II would probably score low in the optimization loop of Article I. Compared to the block sparsity that the algorithms suggest (Figure 7 (a), Article I), the blocks of the restoration project discussed in Article II were built very close to each other (Figure 4 c)). Naturally, the optimization algorithms seek to maximize the effect of each individual block. In that sense, building several blocks successively seems like a bad recipe for optimization, since they will have a redundant effect on WT rise. There is of course a good reason for the suboptimal arrangement of successive blocks in restoration projects. Unlike the theoretical blocks built by the model, blocks in the real world have a possibility to fail under hydraulic pressure (Armstrong et al. 2009; Ritzema et al. 2014). Having successive blocks alleviates the pressure and decreases the risk of failure. A future avenue of research could try to implement this necessary feature into the optimization algorithms used in Article I.

4.2 Impact of water table on the ecosystem

As was shown in Figure 2, there is empirical evidence that peatlands with a shallower average WT emit less CO_2 (Jauhiainen et al. 2012; Carlson et al. 2015; Evans et al. 2022). It then follows from our discussion above that management operations in canals have some, albeit limited, potential to decrease CO_2 emissions in drained peatlands. To have an idea of the magnitude, in the particular peatland area studied in Article II the decrease in CO_2 emissions in the first year after block installation was found to be in the range from $0.62\,\mathrm{Mg\,ha^{-1}}$ to $1.52\,\mathrm{Mg\,ha^{-1}}$, the variation coming from different weather and peat physical properties scenarios[11].

The carbon balance is a crucial axis along which to analyze the state of the peatland

[9] The fraction of CO_2 emissions prevented by the blocks is equal to the fraction of rewetted peat volume. This fact follows from the direct proportionality between the average WT and CO_2 emissions in our models, which is shown in Eq.(1).

[10] To be clear, this fact does not undermine the logic of Article I. The main question of that work is about comparing different blocking locations, and the hydrologic conditions were chosen so that the simple CWL model was sufficient for that purpose.

[11] The coefficients of the linear relationship between WT and CO_2 emissions with which these numbers were calculated were based on the *Acacia* plantations analyzed in Jauhiainen et al. (2012). Although our study site had a more heterogeneous vegetation cover, this CO_2 estimation is likely giving the correct order of magnitude.

ecosystem, but it is not the only one. The results of Article III suggest that nutrient balance is another such axis, at least in forested peatlands. Having some knowledge of the nutrient dynamics gives clarity on what the current and future direction of the peatland vegetation might be. This is particularly true in managed peatland forests, where, in order to understand the ecosystem as a whole, we need to consider humans' economic goals as well as all the rest of the natural processes.

The results of Article III indicate that lowering the CWL from -0.3 m to -0.9 m by performing the usual DNM operations would yield a stand volume growth increase in the range from $0.5\,m^3\,ha^{-1}$ to $3.5\,m^3\,ha^{-1}$ over five years. However, upon investigation of the nutrient balance we found that most stands' growth was limited by K deficiency. This suggests that the causal relationship between lower CWL and enhanced stand growth is, in most sites, modulated by the peat K content, and not only by shallow WTs impeding root metabolism. The connection between the lower CWL—and therefore, WT—and the enhanced growth is therefore indirect. Lower WT increases the rate of peat decomposition, which releases nutrients to be uptaken by plants.

But due to the low concentration of K in the peatlands studied, only 0.29–0.39 kg of K are released for every tonne of peat that gets decomposed (see Table 2 in Article III for nutrient concentrations in the studied sites). This number, according to the empirical equations relating stand volume and nutrient content developed in Palviainen and Finér (2012) and adopted in Article III, is comparable to the amount of K that is required to increase the stand volume by 1 m^3: 0.23–0.33 kg K. Therefore, the business as usual DNM operations of cleaning the ditches in such peatland forests would be a very inefficient way of providing the nutrients they need to grow. Furthermore, lowering the WT through DNM would also have the adverse environmental effect of increasing the already high content of N (and sometimes P) in the peat, which would increase leaching. The same effect in growth could be observed in most stands if K fertilization would be applied while avoiding drainage, which would amount to less CO_2 emissions from the peatland and less N leaching to water courses.

4.3 Process-based models for peatland management

Process-based models, such as the ones developed in this thesis, are based on universal physical principles, and that is perhaps their biggest asset. The laws governing groundwater and open channel flow, peatland nutrient dynamics, and peat creation and subsidence, for instance, are independent of location—the same model that describes groundwater flow in tropical peatlands (Articles I and II) works in boreal peatlands too (Article III). The same physical laws also apply in scenarios for which there is no experimental data. The results of applying different canal block locations (Article I), weather scenarios (Article II), and DNM scenarios (Article III), to name a few, could not have been easily studied using techniques based only on experimental data. The main alternative to process-based models when monitoring and planning management operations in peatlands are statistical models. By construction, and independently of their degree of sophistication, statistical models fail to capture our detailed knowledge about the physical processes involved. In that sense, one could think of process-based models as a tool that enables the extrapolation of measurements outside the training set of experimental data by using physical principles. This is precisely why process-based models are crucial when planning for climate change adaptation: there is no experimental data available about the ecological dynamics of peatland ecosystems in the upcoming extreme climatic conditions.

There are some prices to pay for the universality of physical principles. One of them is the difficulty to parameterize process-based models to describe specific situations. The physical laws underlying process-based models are general mathematical expressions that need to be made concrete to represent real-world phenomena. For instance, the PDEs that govern groundwater flow in peatlands, which under the approximations adopted in this thesis are given by Eq.(11), describe the flow of a fluid in a porous medium. In order for Eq.(11) to approximate the movement of the WT in the particular peatland we are studying—and not of, say, the movement of afternoon coffee inside *pulla*—the right parameters have to be chosen. There are many instances in Articles I–III that may exemplify the point that the parameterization of peatland process-based models is difficult. But let us stick to Eq.(11), which has a central role in this thesis. Of the four parameters of Eq. (11), S_y and T are the most difficult to either measure directly or to infer from other data. Despite being a crucial component of our quantitative understanding of peatland water dynamics, there is a clear and significant gap in our knowledge of their values. As reported in Cobb and Harvey (2019) and cited in Article II, the values of T in tropical peatlands vary in orders of magnitude between studies, between areas in the same site, and vertically within the soil column (Kelly et al. 2014; Baird et al. 2017; Cobb et al. 2017). We tried to solve the parameterization conundrum that this large variability poses by generating a probability distribution for the parameters using Markov Chain Montecarlo (MCMC) methods. Our goal was to find which peat hydraulic properties—S_y and T—generated a WT that corresponded best to the dipwell measurements. MCMC (or any other similar method, for that matter) requires running thousands of simulations in order to search the parameter space of S_y and T. Simulating the WT over the whole area was too expensive computationally. Instead, our attempt consisted in simulating the WT only along the line of the dipwell transects. This simplified Eq.(11) to its one-dimensional version, Eq.(24), and greatly reduced the computational resources needed to run a single simulation. The results, however, were not satisfactory. The posterior distribution of S_y and T was patternless and noisy, meaning that there was no clear way of determining what values of S_y and T best suited the dipwell WT data. We attributed this failure to the water flows in the directions perpendicular to the transect line, which we were ignoring due to the simplification of the groundwater flow equation into one dimension. This parameterization attempt was finally not included in any of the articles of this thesis.

Another price to pay is the difficulty of adjusting the complexity of the model to the task at hand. With process-based models there is a constant trade-off between computational complexity and simulation accuracy. Oftentimes the physical equations yield unpractical models. One important example in this thesis was the description of the interface between WT and CWL in Article II, i.e., the coupling between Eq. (11) and (22). The most accurate simulations would have resulted from fully coupling the equations for the two domains—peatland and canal network—and solving them together (Barthel and Banzhaf 2016). Instead, due to computational constraints, we chose to model the water flow in the two domains separately. This results in some inaccuracies because mass-balance is not strictly preserved at the interface of the domains (Sophocleous 2002).

There are also several rewards for successfully applying process-based models to model peatland processes. Let me highlight two aspects that are particularly important for this thesis: improving the readability of complex processes and enabling scientifically informed decision-making through cost-benefit analyses.

There is no denying that the phenomena described in this thesis resist simple descriptions. Finding the CWL and WT in a large peatland area with varying topography, weather conditions and channel network topology is a difficult task on its own, let alone predicting their interactions with peatland nutrient dynamics, stand merchantable volume, and CO_2 emissions. These are complex, interconnected phenomena, which are very hard to reason about in any other way. If implemented correctly, and despite the many necessary approximations, process-based models have the ability to integrate all these complex factors over large peatland areas, and do so in a way that is amenable for human understanding. In this sense, process-based models provide mock-up representations of the peatland ecosystem which the scientist may prod and tweak in order to explore the underlying mechanisms and their effects.

The other main benefit of process-based models that is relevant to this work is their ability to serve as an aid in practical management scenarios. In each of the studies presented in this thesis, process-based models were used to make comparisons between scenarios that are not feasible using other approaches. Contrasting the relative effectiveness of different block locations in Article I, the weather and hydraulic property scenarios in Article II, and the initial peat nutrient content, peat types and DNM scenarios in Article III, for instance, would require an enormous amount of resources if done by a combination of empirical and statistical methods. And even if those studies were carried out, the relative importance of each variable in the final result would be difficult to separate. In contrast, the variables in process-based models are separable by construction. Once the model is built, it is technically trivial to modify the value of a single parameter and observe its effect on the system. This makes process-based models a very powerful tool to aid stakeholders in making informed plans for peatland management.

4.4 Future directions

There are several ways in which the models presented in each article could be extended and improved.

In order to be more useful to canal-blocking peatland restoration projects, Article I should account for the possibility of dam failure. One practical option could be for the optimization algorithms to propose block locations based on the canal bed slope: the steeper the slope, the more dams should be present.

Article II computed the WT and the associated CO_2 emissions of a single tropical peatland area during one year. One obvious way to increase its scientific relevance would be to study more areas containing different canal network and blocking densities. This work could also be expanded to study the impact of canal blocks in the timescale of decades, which is closer to their lifespan (Ritzema et al. 2014). In this timescale canal blocks would probably have a larger impact on WT, both in terms of magnitude and reach into the peatland.

A number of peatland ecosystem mechanisms are described in Article III using empirically-derived functions rather than process-based models. Upgrading the description of some of those mechanisms to be more process-based could be useful, at least in the cases where doing so would not increase the complexity of the model excessively. A good candidate for this improvement is the organic matter decomposition model. As it is presented in Article III, it does not preserve nutrient mass balance, which is one of the key features that a peatland ecosystem model of this kind should have. Current implementations of the model already have this feature (unpublished). Another improvement for the model, which is related to the previous one, would be to add better accounting of the peatland C fluxes, both as CO_2 emissions to the atmosphere, and as dissolved organic C to the water courses.

The most enticing future research direction does not lie, however, in improving any of the articles on their own, but rather in combining them. A combination of the hydrological model of Article II and a spatially explicit version of the ecosystem model of Article III would enable evaluating the effect of many management practices on a catchment scale. Forested peatland sites are managed with different management targets in mind: forest production, biodiversity restoration, or C capture, for instance. The combination of the models presented in Articles II and III would present a platform to manage this diversity, in which different peatland sites could be devoted to different ecosystem services. Additionally, the more detailed CWL model of Article II could be used in the optimization context of Article I to improve dam location suggestions. By using the Simple Optimization algorithm of Article I (i.e., the genetic algorithm targeting changes in the CWL), this setup could result in an improved algorithm of canal block location optimization.

The models presented in Articles I–III have potential to be impactful outside the academic realm when oriented towards responsible peatland management practices under climate change scenarios. For this potential to be realized, apart from the aforementioned improvements, the role of methane emissions should also be incorporated into the models.

5 CONCLUSIONS

The presented models were found to be useful for comparing different peatland management options with respect to various targets. These models are applicable in a large set of scenarios to answer a variety of practical management questions outside the ones reported in the experiments. Optimization of the canal-block locations significantly improved rewetting results compared to the methods currently used in the practical restoration projects. Our tropical peatland WT simulations revealed previously unreported effects of canal-blocks. The rewetting effect after one year was restricted to a distance of 600 m from the canals, and in some parts of the catchment the blocks lowered WT. Furthermore, the simulations pinpointed the crucial role of precipitation in peatland restoration: When the precipitation was low the blocks were unable to keep the WT at desired levels. This implies that droughts induced by climate change can make the restoration of peatlands exceedingly difficult. In boreal drained peatland forests, WT affected tree growth via organic matter decomposition and nutrient release, which highlights the interactions and feedbacks between the hydrological and biogeochemical processes.

Peatland ecosystems are as complex as they are important. We need tools that help us reason about that complexity, and we need to use them to improve our interactions with these crucial ecosystems.

The scientific consensus seems to be that the next few decades of anthropogenic climate change could be key to our future as a species. Equally certain is the uncertainty in the scenarios we will face. Precipitation, a key component of peatland ecosystems, may increase or decrease to unprecedented levels. Additionally, as the human population increases, more peatland areas are likely to become managed in the coming years. The adaptation measures required to mitigate long-term climate change seem to be misaligned with the short-term economic incentives. Now more than ever, precise information of the consequences of human impact in peatlands is imperative.

In an ideal future, every decision concerned with the management of peatland ecosystems would be grounded in scientific wisdom. The three works presented in this thesis take a humble step towards that future.

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